

# Hydrogeology and Water Quality of a Surficial Aquifer Underlying an Urban Area, Manchester, Connecticut

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# FOREWORD

The mission of the U.S. Geological Survey (USGS) is to assess the quantity and quality of the earth resources of the Nation and to provide information that will assist resource managers and policymakers at Federal, State, and local levels in making sound decisions. Assessment of water-quality conditions and trends is an important part of this overall mission.

One of the greatest challenges faced by water-resources scientists is acquiring reliable information that will guide the use and protection of the Nation's water resources. That challenge is being addressed by Federal, State, interstate, and local water-resource agencies and by many academic institutions. These organizations are collecting water-quality data for a host of purposes that include: compliance with permits and water-supply standards; development of remediation plans for specific contamination problems; operational decisions on industrial, wastewater, or water-supply facilities; and research on factors that affect water quality. An additional need for water-quality information is to provide a basis on which regional- and national-level policy decisions can be based. Wise decisions must be based on sound information. As a society we need to know whether certain types of water-quality problems are isolated or ubiquitous, whether there are significant differences in conditions among regions, whether the conditions are changing over time, and why these conditions change from place to place and over time. The information can be used to help determine the efficacy of existing water-quality policies and to help analysts determine the need for and likely consequences of new policies.

To address these needs, the U.S. Congress appropriated funds in 1986 for the USGS to begin a pilot program in seven project areas to develop and refine the National Water-Quality Assessment (NAWQA) Program. In 1991, the USGS began full implementation of the program. The NAWQA Program builds upon an existing base of water-quality studies of the USGS, as well as those of other Federal, State, and local agencies. The objectives of the NAWQA Program are to:

- Describe current water-quality conditions for a large part of the Nation's freshwater streams, rivers, and aquifers.

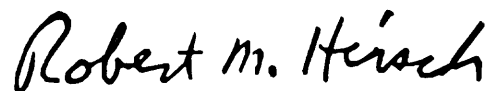
- Describe how water quality is changing over time.
- Improve understanding of the primary natural and human factors that affect water-quality conditions.

This information will help support the development and evaluation of management, regulatory, and monitoring decisions by other Federal, State, and local agencies to protect, use, and enhance water resources.

The goals of the NAWQA Program are being achieved through ongoing and proposed investigations of 59 of the Nation's most important river basins and aquifer systems, which are referred to as study units. These study units are distributed throughout the Nation and cover a diversity of hydrogeologic settings. More than two-thirds of the Nation's freshwater use occurs within the 59 study units and more than two-thirds of the people served by public water-supply systems live within their boundaries.

National synthesis of data analysis, based on aggregation of comparable information obtained from the study units, is a major component of the program. This effort focuses on selected water-quality topics using nationally consistent information. Comparative studies will explain differences and similarities in observed water-quality conditions among study areas and will identify changes and trends and their causes. The first topics addressed by the national synthesis are pesticides, nutrients, volatile organic compounds, and aquatic biology. Discussions on these and other water-quality topics will be published in periodic summaries of the quality of the Nation's ground and surface water as the information becomes available.

This report is an element of the comprehensive body of information developed as part of the NAWQA Program. The program depends heavily on the advice, cooperation, and information from many Federal, State, interstate, Tribal, and local agencies and the public. The assistance and suggestions of all are greatly appreciated.



Robert M. Hirsch  
Chief Hydrologist



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## CONVERSION FACTORS AND VERTICAL DATUM

### CONVERSION FACTORS

Multiply	By	To obtain
<b>Length</b>		
inch (in.)	2.54	centimeter
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
<b>Area</b>		
square mile (mi <sup>2</sup> )	259.0	hectare
square mile (mi <sup>2</sup> )	2.590	square kilometer
<b>Volume</b>		
million gallons (Mgal)	3,785	cubic meter
cubic foot (ft <sup>3</sup> )	0.02832	cubic meter
<b>Flow rate</b>		
inch per year (in/yr)	25.4	millimeter per year
cubic foot per second ((ft <sup>3</sup> /s)	0.02832	cubic meter per second
<b>Hydraulic conductivity</b>		
foot per day (ft/d)	0.3048	meter per day
<b>Transmissivity*</b>		
foot squared per day (ft <sup>2</sup> /d)	0.09290	meter squared per day
<b>Mass</b>		
pound per day (lb/d)	0.4536	kilogram per day

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$^{\circ}\text{F} = (1.8 \times ^{\circ}\text{C}) + 32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$^{\circ}\text{C} = (^{\circ}\text{F} - 32) / 1.8$$

**Altitude**, as used in this report, refers to distance above or below sea level.

**\*Transmissivity:** The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft<sup>3</sup>/d)/ft<sup>2</sup>]ft. In this report, the mathematically reduced form, foot squared per day (ft<sup>2</sup>/d), is used for convenience.

**Specific conductance** is given in microsiemens per centimeter at 25 degrees Celsius (μS/cm at 25 °C).

**Concentrations of chemical constituents** in water are given either in milligrams per liter (mg/L) or micrograms per liter (μg/L).

### VERTICAL DATUM

**Sea level:** In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

# Hydrogeology and Water Quality of a Surficial Aquifer Underlying an Urban Area, Manchester, Connecticut

By John R. Mullaney and Stephen J. Grady

## Abstract

The quality of water along flowpaths in a surficial aquifer system in Manchester, Connecticut, was studied during 1993-95 as part of the National Water Quality Assessment program. The flowpath study examined the relations among hydrogeology, land-use patterns, and the presence of contaminants in a surficial aquifer in an urban area, and evaluated ground water as a source of contamination to surface water.

A two-dimensional, finite-difference ground-water-flow model was used to estimate travel distance, which ranged from about 50 to 11,000 feet, from the source areas to the sampled observation wells. Land use, land cover, and population density were determined in the source areas delineated by the ground-water-flow simulation. Source areas to the wells contained either high- or medium-density residential areas, and population density ranged from 629 to 8,895 people per square mile.

Concentrations of selected inorganic constituents, including sodium, chloride, and nitrite plus nitrate nitrogen, were higher in the flowpath study wells than in wells in undeveloped areas with similar aquifer materials. One or more of 9 volatile organic compounds were detected at 12 of 14 wells. The three most commonly detected volatile organic compounds were chloroform, methyl-*tert*-butyl ether, and trichloroethene. Trichloroethene was detected at concentrations greater than the maximum contaminant level for drinking water (5 micrograms per liter) in samples from one well. Four pesticides, including dichloro diphenyl dichloroethylene, dieldrin, dichloroprop, and simazine were detected at low concentrations.

Concentrations of sodium and chloride were higher in samples collected from wells screened in the top of the saturated zone than in samples collected from deeper zones. Volatile organic compounds and elevated concentrations of nitrite plus nitrate as nitrogen were detected at depths of as much as 60 feet

below the water table, indicating that the effects of human activities on the ground-water quality extends to the bottom of the surficial aquifer.

The age of ground water, as determined by tritium and <sup>3</sup>helium concentrations, was 0.9 to 22.6 years. pH, alkalinity, and calcium were higher and concentrations of dissolved oxygen were lower in ground-water samples with ages of 10 years or more than in samples younger than 10 years. In addition, concentrations of sodium, chloride, and nitrite plus nitrate nitrogen were low in ground-water samples with ages of 10 years or more, indicating that concentrations of these compounds may be increasing with time or that the recharge areas to these wells may have had less intensive urban land use. Methyl-*tert*-butyl ether was detected only in wells with ground water ages of less than 11 years, which is consistent with the date of introduction of this compound as a gasoline additive in Connecticut.

Analysis of additional samples collected for analysis of stable nitrogen isotopes indicated that the most likely source of elevated concentrations of nitrate nitrogen was lawn and garden fertilizers, but other sources, including wastewater effluents, soil organic nitrogen, and atmospheric deposition, may contribute to the total. Population density was positively correlated (at the 97 percent confidence level) to concentrations of nitrite plus nitrate as nitrogen.

Water quality in the Hockanum River aquifer has been degraded by human activities, and, after discharge to surface water, affects the water quality in the Hockanum River. On an annual basis, ground-water discharge from the study area to the river (as measured at a downstream continuous-record gaging station) contributes about 5 percent of the annual load of nitrite plus nitrate nitrogen, but, during low flow, contributes 11 percent of the nitrite plus nitrate nitrogen, 32 percent of the calcium, and 16 percent of the chloride to the river.



## INTRODUCTION

Effects of urban land use on ground-water quality are a major concern in many parts of the Nation. Many cities and towns have concerns about the degradation of ground water used for drinking-water supplies and about how degraded ground water affects surface-water quality and ecological and recreational resources. Point-source ground-water contamination in urban areas has been attributed to leaky underground storage tanks, chemical spills, leaky sewers, uncovered storage areas for road salt, and other sources. Little information has been collected on nonpoint-source contaminants in ground water or on contaminants that discharge to surface waters from aquifers underlying urban areas.

In 1991, the U.S. Geological Survey (USGS) began implementation of the National Water-Quality Assessment (NAWQA) program. The long-term goals of the NAWQA program are to describe the status and trends in the quality of a large, representative part of the Nation's surface- and ground-water resources and to provide a sound scientific understanding of the primary natural and human factors affecting the quality of these resources (Leahy and others, 1990). A major component of the NAWQA program is being accomplished by investigating large river basins (study units). Ground-water studies conducted in each study unit as part of the NAWQA program have been categorized as follows:

(1) Study-unit surveys, designed to provide a broad overview of ground-water quality in each large river basin.

(2) Land-use studies, designed to assess the quality of recently recharged, shallow ground water associated with regionally extensive combinations of land use and hydrologic conditions.

(3) Flowpath studies, the focus of this report, designed to (1) characterize the spatial and temporal distribution of water quality in shallow ground-water systems for particular settings, (2) increase understanding in these settings of the natural processes and anthropogenic factors that control the evolution of ground-water quality along flowpaths through the saturated zone, and (3) compare shallow ground-water quality and stream-water quality under baseflow conditions. Sites for flowpath studies were selected in a setting that was sampled as part of a land-use study to integrate other aspects of the NAWQA program. Sites also were selected to coincide with subbasins where

NAWQA surface-water samples were collected to indicate water-quality effects of selected land uses (Gilliom and others, 1995).

The flowpath study described in this report was conducted as part of the NAWQA program in the Connecticut, Housatonic, and Thames River Basins study unit. The population of 59 cities in the study unit is 20,000 people or more. These cities contain about 60 percent of the population of these basins. Even in the smaller towns, urban centers may have small sections of densely populated or industrialized land. The Hockanum River aquifer in Manchester, Connecticut, was selected for a flowpath study because the area contains many of the required criteria, and there was some existing information on hydrogeology and ground-water quality.

## Purpose and Scope

This report presents the results of a flowpath study in Manchester, Connecticut, to examine relations among hydrogeology, land-use patterns, and the presence of contaminants in the surficial aquifer system, and to evaluate ground water as a source of contamination to surface water. Information also is included on the hydrogeology of the surficial aquifer, spatial and temporal variations in chemical quality of ground water, direction and magnitude of ground-water flow, age of ground water, and the amount of water withdrawn from the surficial aquifer for public supply. A two-dimensional, steady-state, finite-difference ground-water-flow model was used to improve the understanding of the ground-water system in the Manchester area and to estimate the travel distances from source areas to sampled wells.

## Previous Investigations

The quality of water in the Hockanum River aquifer was most recently described by Grady (1994). Water samples were collected from 18 water-table wells installed in the stratified-drift aquifer in Manchester during 1987-88 and used, in conjunction with samples from similar wells in other parts of Connecticut, as part of a statistical study to determine the effects of land use on shallow ground-water quality. Analysis of the data indicated that ground-water quality in Manchester is affected by human activities.



Concentrations of several inorganic constituents were elevated when compared with concentrations in undeveloped areas, and several pesticides and volatile organic compounds (VOCs) were detected at low concentrations. Other studies in the Manchester area have included information on the hydrogeology (Gregory and Ellis, 1916; Ryder and others, 1981), bedrock altitude (Colton and Cushman, 1963; Ryder, 1972), and surficial geology (Colton, 1965; Stone and others, 1992).

## Description of Study Area

The Connecticut, Housatonic and Thames River Basins study unit is about 15,750 mi<sup>2</sup>; it extends from the Canadian border to Long Island Sound and includes coastal basins in Connecticut. The area studied for this flowpath investigation includes most of Manchester, a small city of about 52,000 people in central Connecticut (fig. 1). Most of the city is underlain by stratified-drift deposits (the Hockanum River aquifer) overlying arkosic sedimentary bedrock. The eastern end of the city is underlain by crystalline metamorphic bedrock, with altitudes as much as 750 ft. Land-surface altitude of the Hockanum River aquifer in the study area ranges from 70 to about 350 ft.

Median annual precipitation at nearby Bradley International Airport in Windsor Locks for 1961-90 was 43.12 in. (Owenby and Ezell, 1992). Precipitation is distributed evenly throughout the year, but because of evapotranspiration, ground-water recharge takes place primarily during the nongrowing season from October to April. In urban areas, such as Manchester, some recharge may be derived from other sources of water, including leaky water and sewer mains and lawn irrigation.

Land use in Manchester is a mixture of medium- and high-density residential areas, and commercial and forested areas (fig. 2). Although not shown in figure 2, there also are industrial areas, and areas with low-density residential land use. Most of the city is sewered, and only 8.1 percent of households use on-site septic systems. Public water supply is a mixture of ground-water sources and surface-water reservoirs and is provided to 95 percent of households (U.S. Department of Commerce, 1991). Ground water is withdrawn for public supply from seven wells in the stratified-drift aquifer (fig. 1) and two wells in the arkosic bedrock aquifer.

## Acknowledgments

We would like to acknowledge the cooperation, information, and assistance provided by officials from the town of Manchester, Connecticut, East Catholic High School, and by private landowners, who permitted sampling on their property. We would also like to thank Peter Schlosser and Millie Klas of the Lamont-Doherty Earth Observatory of Columbia University for their efforts in dating the ground-water samples; and our U.S. Geological Survey colleagues—Christopher R. Hudon and Kevin L. Vanderveer for their assistance in collecting and interpreting data used in this report, and J. Jeffrey Starn for his technical assistance on ground-water modeling.

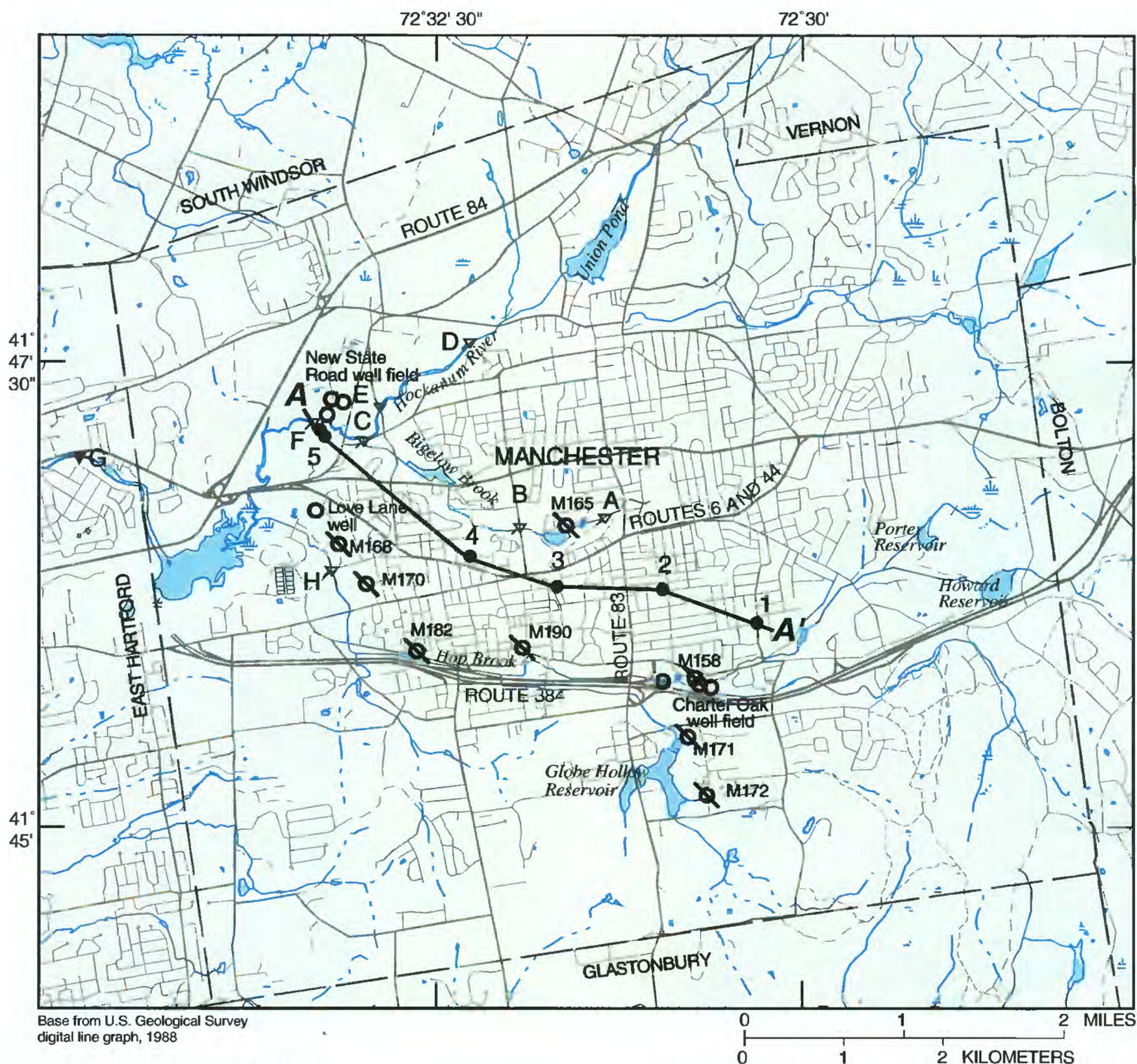
## DATA COLLECTION AND ANALYSIS

The flowpath study was conducted in Manchester during 1993-95. The approach used consisted of the following steps: compiling existing information, selecting locations for well installation, drilling and sampling the wells, constructing a ground-water-flow model, and interpreting the water-quality data.

### Site Selection and Well Installation

A preliminary contour map of the water table was constructed from existing data and used to locate possible flowpaths that could be studied. An area was selected that approximated the drainage divide between two streams, Bigelow and Hop Brooks (fig. 1). Sites for well clusters along the path of ground-water flow were selected in the field by examining upgradient land use and by looking for areas that had access for drilling. In Manchester, little open space is present because of the housing density; therefore, sites were chosen at schools, vacant lots, and on municipal property. Five sites were selected—a single well was installed at site 1 and clusters of three wells were installed at sites 2 to 5 (fig. 1). In addition, one streambed well was installed in the Hockanum River near site 5. The farthest upgradient well was installed near the eastern end of the aquifer, and the farthest downgradient well was near the Hockanum River at the western end of Manchester.





Connecticut, Housatonic, and Thames River Basins



#### EXPLANATION

- A—A'** LINE OF HYDROGEOLOGIC SECTION - (section shown in figure 5)
- 1** FLOWPATH GROUND-WATER QUALITY SITE AND NUMBER
- GROUND-WATER LEVEL OBSERVATION WELL
- PUBLIC SUPPLY WELL TAPPING STRATIFIED DRIFT AQUIFER
- CONTINUOUS STREAMFLOW-GAGING STATION AND NAWQA MONTHLY WATER-QUALITY SITE
- LOW-FLOW SURFACE-WATER SITES
- Water-quality site with discharge measurements
- Discharge measurement site

**Figure 1.** Location of the Manchester study area and data-collection sites for the flowpath study in the Connecticut, Housatonic, and Thames River Basins.



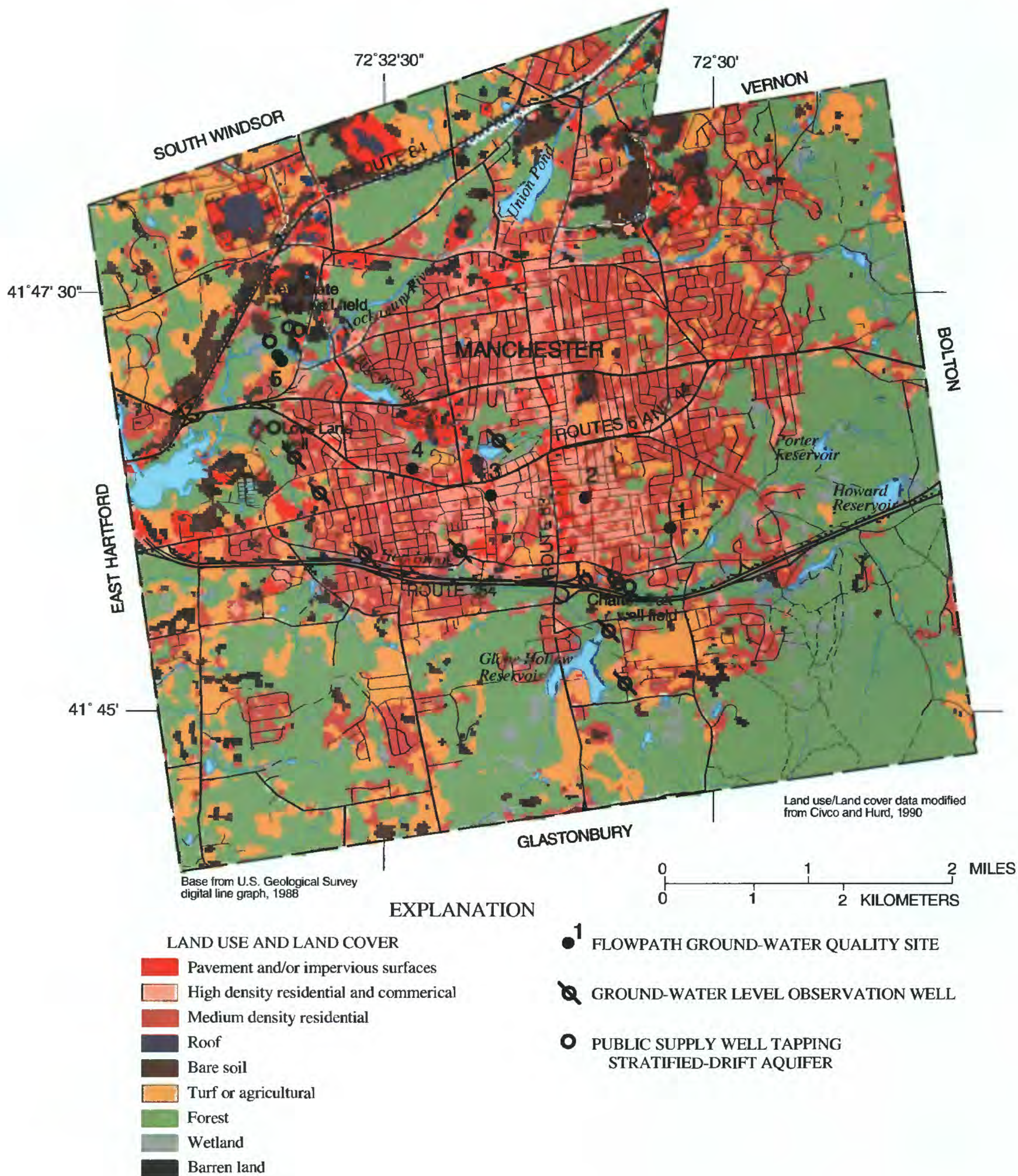


Figure 2. Land use and land cover in Manchester, Connecticut, 1990.



Flowpath wells were installed at each location with a hollow-stem auger drill rig using methods described by Lapham and others (1995). At sites where multiple wells were installed, a test hole was first drilled to bedrock to determine the saturated thickness, and split-spoon samples were collected every 5 to 10 ft to determine lithology. For each cluster, three wells were screened at different depths—one near the bottom of the aquifer, one in the middle of the aquifer, and one near the water table. Wells were constructed of pre-cleaned 2-inch inside diameter schedule 40 polyvinyl chloride (PVC) casing and 5-foot slotted screens. Bentonite was used to seal the annulus between the casing and the borehole, and small diameter manhole covers or locking steel standpipes were cemented in place at the land surface.

The streambed well was a 2-inch inside diameter stainless-steel well with a 2-foot screen that was installed in the upgradient edge of the Hockanum River. This well was installed with a sledge hammer and was driven to a depth 3 ft below the streambed.

## Water-Level Measurements and Water-Quality Samples

Except for the streambed well, water levels in all flowpath wells were measured on a weekly basis. Water levels also were measured monthly in an additional nine existing wells in Manchester to monitor water-table fluctuations in different parts of the aquifer and to provide data for model calibration.

Water samples were collected in the study area during 1994-95 using methods described by Koterba and others (1995). Water samples were collected as many as 7 times for each group of constituents from the 14 wells; samples were analyzed for field parameters and concentrations of major ions, nutrients, trace elements, pesticides, and VOCs. Complete lists of all water-quality variables for which samples were analyzed are shown in tables 1, 2, and 3.

Surface-water samples were collected at low flow at six locations (fig. 1) using a point-sample method. These samples were collected at low flow for comparison with ground-water samples. All low-flow samples were analyzed for concentrations of major ions and nutrients by the USGS National Water Quality Laboratory in Arvada, Colorado.

**Table 1.** Field measurements, inorganic constituents, and carbon measured in the field or analyzed at the U.S. Geological Survey National Water Quality Laboratory for ground-water samples from flowpath study wells, Manchester, Connecticut

[Field sample collection and measurement protocols and laboratory analytical methods are described by Wershaw and others, 1987; Fishman and Friedman, 1989; Fishman, 1993; Koterba and others, 1995; Lapham and others, 1995.  $\mu\text{S}/\text{cm}$  at 25 C, microsiemen per centimeter at 25 degrees Celsius;  $\text{mg}/\text{L}$ , milligram per liter;  $\mu\text{g}/\text{L}$ , microgram per liter; na, not applicable]

Property or constituent	Units	Reporting limit(s)
<b>FIELD MEASUREMENTS</b>		
Specific conductance .....	$\mu\text{S}/\text{cm}$ at 25°C	
pH .....	standard units	na
Temperature .....	°C	na
Dissolved oxygen .....	$\text{mg}/\text{L}$	na
Alkalinity as $\text{CaCO}_3$ .....	$\text{mg}/\text{L}$	na
Bicarbonate .....	$\text{mg}/\text{L}$	na
<b>LABORATORY DETERMINATIONS</b>		
Hardness as $\text{CaCO}_3$ .....	$\text{mg}/\text{L}$	na
Noncarbonate hardness .....	$\text{mg}/\text{L}$	na
Calcium .....	$\text{mg}/\text{L}$	na
Magnesium .....	$\text{mg}/\text{L}$	na
Sodium .....	$\text{mg}/\text{L}$	na
Potassium .....	$\text{mg}/\text{L}$	na
Sulfate .....	$\text{mg}/\text{L}$	na
Chloride .....	$\text{mg}/\text{L}$	na
Fluoride .....	$\text{mg}/\text{L}$	0.1
Bromide .....	$\text{mg}/\text{L}$	0.01
Silica .....	$\text{mg}/\text{L}$	na
Dissolved solids, residue at 180 °C .....	$\text{mg}/\text{L}$	na
Nitrite as N .....	$\text{mg}/\text{L}$	0.01
Nitrite plus nitrate as N .....	$\text{mg}/\text{L}$	0.05
Nitrogen, ammonia as N .....	$\text{mg}/\text{L}$	0.01 - 0.02
Nitrogen, ammonia plus organic as N .....	$\text{mg}/\text{L}$	0.2
Phosphorus as P .....	$\text{mg}/\text{L}$	0.01
Orthophosphate as P .....	$\text{mg}/\text{L}$	0.01
Iron .....	$\mu\text{g}/\text{L}$	3.0
Manganese .....	$\mu\text{g}/\text{L}$	1.0
Arsenic .....	$\mu\text{g}/\text{L}$	1.0
Barium .....	$\mu\text{g}/\text{L}$	1.0
Beryllium .....	$\mu\text{g}/\text{L}$	1.0
Cadmium .....	$\mu\text{g}/\text{L}$	1.0
Chromium .....	$\mu\text{g}/\text{L}$	1.0
Cobalt .....	$\mu\text{g}/\text{L}$	1.0
Copper .....	$\mu\text{g}/\text{L}$	1.0
Lead .....	$\mu\text{g}/\text{L}$	1.0
Molybdenum .....	$\mu\text{g}/\text{L}$	1.0
Nickel .....	$\mu\text{g}/\text{L}$	1.0
Silver .....	$\mu\text{g}/\text{L}$	1.0
Zinc .....	$\mu\text{g}/\text{L}$	1.0
Antimony .....	$\mu\text{g}/\text{L}$	1.0
Aluminum .....	$\mu\text{g}/\text{L}$	1.0
Selenium .....	$\mu\text{g}/\text{L}$	1.0
Uranium .....	$\mu\text{g}/\text{L}$	1.0
Carbon, dissolved, organic as C .....	$\text{mg}/\text{L}$	na



**Table 2.** Pesticide compounds analyzed for in water samples from flowpath study wells, Manchester, Connecticut

[All pesticide analyses were performed by the U.S. Geological Survey National Water Quality Laboratory by solid-phase extraction and capillary-column gas chromatography/mass spectrometry (Lindley and others, 1994; Zaugg and others, 1995) or by high-performance liquid chromatography (Werner and others, 1996). Method detection limits (MDL) and minimum reporting limits (MRL) are compound specific and expressed in micrograms per liter. Use: F, fungicide; H, herbicide; I, insecticide; M, metabolite]

Pesticide compound (common chemical name)	MDL (MRL if different)	Use	Pesticide compound (common chemical name)	MDL (MRL if different)	Use
Acetochlor.....	0.002	H	Fluometuron.....	0.035	H
Acifluorfen (Blazer) .....	.035	H	Fonofos (Dyfonate).....	.003	I
Alachlor (Lasso).....	.002	H	HCH, <i>alpha</i> -.....	.002	I
Aldicarb (Temik).....	.016	I	HCH, <i>gamma</i> - (Lindane) .....	.004	I
Aldicarb sulfone.....	.016	M	Linuron .....	.002	H
Aldicarb sufoxide.....	.021	M	Malathion .....	.005	I
Atrazine.....	*.001	H	MCPA (Metaxon) .....	.050	H
Atrazine, desethyl.....	.001	M	MCPB (Tropotox) .....	.035	H
Azimphos, methyl (Guthion) .....	**0.001	I	Methiocarb (Mesurol) .....	.026	I
Benfluralin (Benefin).....	.002	H	Methomyl (Lannate) .....	.017	I
Bentazon (Basagran).....	.014	H	Metolachlor (Dual) .....	*.002	H
Bromacil (Bromax) .....	.035	H	Metribuzin (Sencor) .....	.004	H
Bromoxynil (Torch) .....	.035	H	Molinate (Ordram).....	.004	H
Butylate (Genate plus) .....	.002	H	Napropamide (Devrinol).....	.003	H
Carbaryl (Sevin).....	**0.003	I	1-Naphthol (Alpha Naphthol) .....	.007	M
Carbofuran (Furadan).....	**0.003	I	Neburon (Neberex) .....	.015	H
3-OH-Carbofuran.....	.014	M	Norflurazon (Telok).....	.024	H
Chloramben (Amiben) .....	.011	H	Oryzalin (Surflan) .....	.019	H
Chlorothalonil (Bravo) .....	.035	F	Oxamyl (Vydate).....	.018	I
Chlorpyrifos (Lorsban) .....	.004	I	Parathion, ethyl .....	.004	I
Clopyralid (Stinger) .....	.050	H	Parathion, methyl .....	.006	I
Cyanazine.....	.004	H	Pebulate (Tillam).....	.004	H
Dacthal (DCPA) .....	.002	H	Pendimethalin (Prowl) .....	.004	H
Dacthal, mono-acid.....	.017	M	Permethrin, <i>cis</i> .....	.005	I
Diazinon.....	.002	I	Phorate (Thimet) .....	.002	I
Dicamba (Banval) .....	.035	H	Picloram (Amdon).....	.050	H
Dichlobenil.....	.020	H	Prometon .....	.018	H
Dichlorodiphenyldichloroethylene (p,p'-DDE).....	.006	M	Pronamide (Kerb).....	.003	H
	*(.001)		Propachlor (Ramrod).....	.007	H
2,4-Dichlorophenoxyacetic acid (2,4-D) .....	.035	H	Propanil (Stampede).....	.004	H
4-(2,4-Dichlorophenoxy) butyric acid (2,4-DB) .....	.035	H	Propargite (Omite) .....	.013	I
Dichlorprop (2,4-DP) .....	.032	H	Propham (IPC) .....	.035	H
Dieldrin .....	.001	I	Propoxur (Baygon).....	.035	I
2,6-Diethylalanine.....	.003	M	Silvex (2,4,5-TP) .....	.021	H
4,6-Dinitro-o-cresol (DNOC).....	.035	H,I	Simazine (Princep).....	.005	H
Dinoseb (DNPB).....	.035	H,I	Tebuthiuron (Spike) .....	.010	H
Disulfoton .....	.017	I	Terbacil (Sinbar) .....	**0.007	H
Diuron (DCMU).....	.020	H	Terbufos (Counter).....	.013	I
EPTC.....	.002	H	Thiobencarb (Bolero).....	.002	H
Ethalfuralin (Sonalan) .....	.004	H	Triallate (Far-Go) .....	.001	H
Ethoprop (Mocap).....	.003	I	2,4,5-Trichlorophenoxy-acetic acid (2,4,5-T) .....	.035	H
Esfenvalerate (Asana) .....	.019	I	Triclopyr (Crossbow) .....	.050	H
Fenuron (Beet-Klean) .....	.013	H	Trifluralin (Treflan) .....	.002	H

\*Some reported values are estimated for concentrations at or less than the MDL or greater than the calibration range.

\*\*All reported values are estimates due to problems with gas chromatography or extraction.

**Table 3.** Volatile organic compounds analyzed for in water samples from flowpath study wells, Manchester, Connecticut

[All volatile organic compound analyses were performed by the U.S. Geological Survey National Water Quality Laboratory using purge and trap capillary gas chromatography/mass spectrometry methods (Raese and others, 1995; Rose and Schroeder, 1995). The reporting limit for all volatile organic compounds is 0.2 µg/L (microgram per liter) except for dibromochloropropane, which has a reporting limit of 1.0 µg/L]

Volatile organic compound		
Benzene	1,2-Dichlorobenzene	Methyl <i>tert</i> -butyl ether
Bromobenzene	1,3-Dichlorobenzene	Naphthalene
Bromochloromethane	1,4-Dichlorobenzene	<i>n</i> -Propylbenzene
Bromodichloromethane	Dichlorodifluoromethane	Styrene
Bromoform	1,1-Dichloroethane	1,1,1,2-Tetrachloroethane
Bromomethane	1,2-Dichloroethane	1,1,2,2-Tetrachloroethane
<i>n</i> -Butylbenzene	1,1-Dichloroethene	Tetrachloroethene
<i>sec</i> -Butylbenzene	<i>cis</i> -1,2-Dichloroethene	Toluene
<i>tert</i> -Butylbenzene	<i>trans</i> -1,2-Dichloroethene	1,2,3-Trichlorobenzene
Carbon tetrachloride	1,2-Dichloropropane	1,2,4-Trichlorobenzene
Chlorobenzene	1,3-Dichloropropane	1,1,1-Trichloroethane
Chloroethane	2,2-Dichloropropane	1,1,2-Trichloroethane
Chloroform	1,1-Dichloropropene	Trichloroethene
Chloromethane	<i>cis</i> -1,3-Dichloropropene	Trichlorofluoromethane
2-Chlorotoluene	<i>trans</i> -1,3-Dichloropropene	1,2,3-Trichloropropane
4-Chlorotoluene	Ethylbenzene	Trichlorotrifluoroethane
Dibromochloromethane	Hexachlorobutadiene	1,2,4-Trimethylbenzene
Dibromochloropropane	Isopropylbenzene	1,3,5-Trimethylbenzene
1,2-Dibromoethane	<i>p</i> -Isopropyltoluene	Vinyl chloride
Dibromomethane	Methylene chloride	Xylenes ( <i>meta</i> -, <i>para</i> -, <i>ortho</i> -)

Ground-water samples were collected at different times during a 2-year period (fig. 3) to determine whether there was any seasonal or temporal variability in concentrations of selected constituents.

Surface-water samples (collected as part of the NAWQA surface-water-quality studies) were collected monthly from March 1993 to September 1995 at a USGS streamflow-gaging station (Hockanum River near Manchester, station 01192500), about 1.5 mi downstream from the study area (station G, fig. 1).

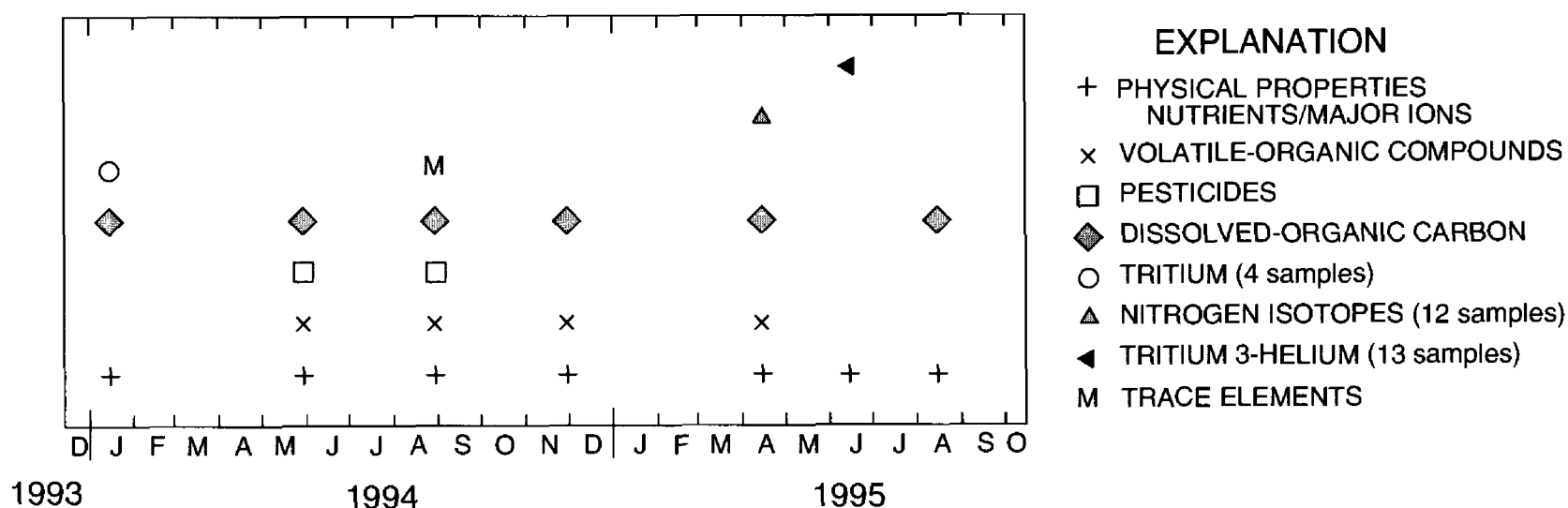
## Quality-Assurance/Quality-Control Procedures

In addition to the environmental samples, 35 quality-assurance/quality-control (QA/QC) samples also were collected at 8 flowpath-study wells. The QA/QC samples collected during the study included 30 field blanks and 5 environmental replicates. Blanks were analyzed for inorganic constituents, trace elements, dissolved organic carbon (DOC),

pesticides, and VOCs. Environmental replicates were analyzed for inorganic constituents, trace elements, DOC, and VOCs.

Field blanks indicated that, for most constituents, ground-water samples were not contaminated from any systematic source, such as cross contamination between wells or from routine cleaning and handling of sampling equipment, supplies, or samples). QA/QC samples documented a widespread problem with three VOC compounds—1,2-dichloroethane, chloroethane, and chloromethane (methyl chloride)—associated with the preservation of NAWQA VOC samples with hydrochloric acid. Systematic contamination with these compounds was documented in this and other NAWQA studies during 1993-95. All data for these compounds have been noted in the database and are not used in this or other NAWQA reports.

Concern about the validity of data for one additional analyte is warranted. QA/QC samples for DOC show that concentrations in the field blanks are not significantly different from DOC concentrations in the environmental samples. Contamination of field blanks was likely due to the use of detergents and methanol to clean the sampling equipment. Even a small residue of methanol on the equipment after cleaning could produce the concentrations observed in the field blanks and complicate interpretation of the environmental data. DOC concentrations in ground-water samples varied greatly from site to site but were relatively consistent for sites where DOC resampling was conducted. Although the process of collecting a ground-water sample uses the same cleaning procedures as those used before collecting field blanks, ground-water sample collection normally involves flushing the sample pump with much greater volumes of native water than the amount of rinse water used prior to taking a field blank. DOC concentrations in the field blanks possibly are an artifact of this unavoidable difference in the processing of blanks and environmental samples. If so, the DOC concentrations in the environmental



**Figure 3.** Dates and types of analyses of ground-water samples collected in Manchester, Connecticut, 1994-95.

samples would represent actual conditions in the aquifer or a combination of native DOC plus some variable input of DOC from pre-sample cleaning. Because it is impossible to determine at this time just what the DOC data for the ground-water samples represent, all data have been marked as suspect, but DOC concentrations for some ground-water samples are presented in this report.

Three inorganic constituents (nitrogen-ammonia as N, phosphorous as P, and calcium) were detected in a few of the field blanks. The nitrogen-ammonia as N and phosphorous as P (0.01 to 0.02 mg/L) were detected at concentrations equal to or slightly greater than the reporting limits, and many of the environmental samples had similar concentrations. Some low-level detections of ammonia and phosphorous, and a few of the other inorganic constituents detected in field blanks, possibly are from concentrations present in the inorganic blank water itself. Analytical results for QA/QC samples from some lots of water used during this study have reported detections of nitrogen-ammonia as N, phosphorus as P, calcium, potassium, sodium, chloride, and silica at concentrations approaching those measured in some field blanks. Concentrations of aluminum were detected in all three field blanks collected for trace elements; therefore, data on aluminum are not discussed further in this report.

Analysis of replicate samples showed that, for the most part, concentrations from sequential samples were reproducible. One exception occurred when replicates were collected for VOCs. The concentration

of trichloroethene in one replicate sample was twice the concentration in the original sample. In addition, two compounds not detected in the original samples, or in any previous samples at this site—carbon tetrachloride and 1,1,1 trichloroethane—were detected at low concentrations. One possible explanation for the discrepancy is that the sample required a larger than normal amount of hydrochloric acid preservative. Consequently, the sample had to be diluted. The replicate possibly was not diluted at the lab, but the corrections were made to the concentrations on the basis of the dilution factor. There is a high degree of confidence in the concentrations of VOCs measured in this investigation because of the reproducibility of low-level concentrations of VOCs from different time periods at the same wells.

## Ground-Water Samples for Age Determinations

Samples were collected in January 1994 to determine if detectable levels of tritium ( $^3\text{H}$ ) were present in ground water and to indicate whether ground water was recharged before or after nuclear testing during the 1960s (Plummer and others, 1993). Additional samples were collected in June 1995 for tritium and  $^3\text{H}$ - $^3\text{He}$  ( $^3\text{H}$ - $^3\text{He}$ ) at selected wells to determine age of ground water. Age determination of the ground-water samples was done by the Lamont-Doherty Earth Observatory of Columbia University using methods described by Schlosser (1992).



Ground-water ages can be used to help identify the introduction and residence time of a solute in ground water and are useful in the calibration of ground-water-flow models. The age-determination method is based on separating out the tritiogenic  $^3\text{He}$  (that produced by the radioactive decay of tritium) component in the water sample and calculating the time elapsed during radioactive decay. This is done using the equation from Plummer and others (1993):

$$t = \frac{t_{1/2}}{\ln 2} \ln \left[ 1 + \frac{{}^3\text{He}_{\text{trit}}}{{}^3\text{H}} \right], \quad (1)$$

where,

$t$  is the time that the sample has been isolated from the atmosphere,

$t_{1/2}$  is the half life of  $^3\text{H}$  or about 12.43 years, and

${}^3\text{He}_{\text{trit}}$  is the  $^3\text{He}$  derived from radioactive decay of tritium, or tritiogenic  $^3\text{He}$ .

Other sources of  $^3\text{He}$  must be separated from  ${}^3\text{He}_{\text{trit}}$  before an age can be determined. This is done using noble gas measurements from the sample (Plummer and others, 1993). Some uncertainties are associated with these measurements. The first of these is in separating the  ${}^3\text{He}_{\text{trit}}$ . This uncertainty is reported to be  $\pm 0.5$  year. Because the age is determined from the time that the water was isolated from the atmosphere, the method does not account for the time the water was in the unsaturated zone or at the surface of the water table, where it might be able to interact with the atmosphere and  ${}^3\text{He}_{\text{trit}}$  could be lost from the water. Other possible errors associated with the measurement include the effects of hydrodynamic dispersion, which generally will make the water appear to be older than its true advective age for waters recharged after 1964. However, because atmospheric concentrations of tritium have been similar for the last 20 to 25 years, the effects of dispersion should be minimal. Solomon and Sudicky (1991) suggest that for ground-water samples recharged after 1975, the age discrepancy should be no more than 10 percent in permeable unconfined aquifers. More errors can be introduced through the sampling process when waters of different recharge ages are integrated or averaged over the entire length of the screened interval. This also could occur in the aquifer at locations where waters may mix as they move upward toward a discharge area.

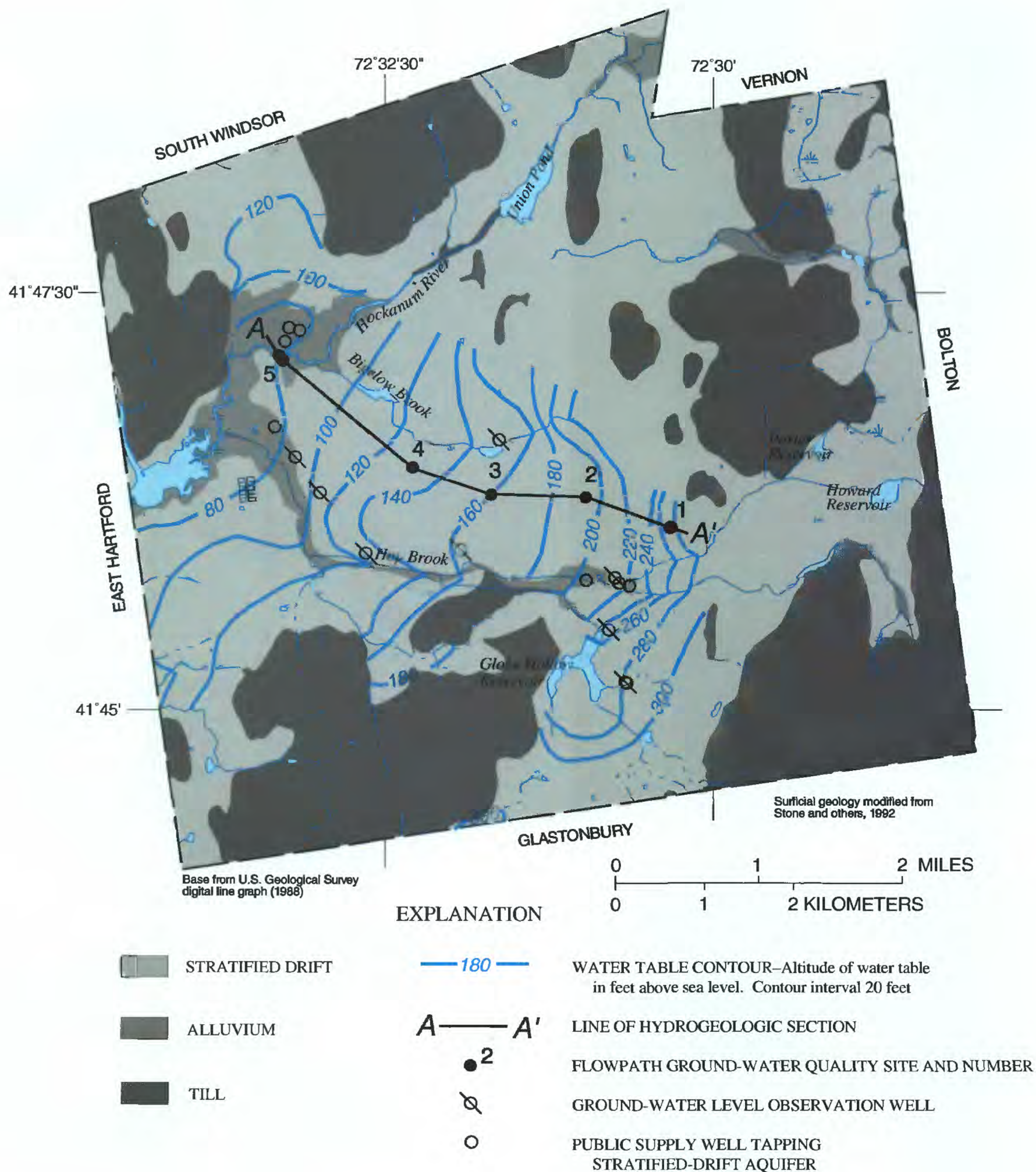
## Ground-Water Samples for Nitrogen Isotope Ratios

The ratio ( $\delta^{15}\text{N}$ ) of the stable isotopes nitrogen-15 ( $^{15}\text{N}$ ) to nitrogen-14 ( $^{14}\text{N}$ ) of the nitrogen in nitrate was measured in 12 samples to assist in determining the source of nitrate nitrogen. This technique has been used in recent years primarily to distinguish between sources of nitrogen that are derived from human and (or) animal waste and those derived from other sources (Kendall and others, 1995). Human and (or) animal waste sources generally have  $\delta^{15}\text{N}$  values of +10 to +20 per mil in water samples containing nitrate, if no denitrification has occurred. Atmospheric and fertilizer sources (because nitrogen fertilizer is produced from atmospheric nitrogen), and soil-derived nitrate have overlapping compositions that generally range from -4 to +9 per mil (Heaton, 1986).

## HYDROGEOLOGY OF THE SURFICIAL AQUIFER

A large part of Manchester is underlain by unconsolidated Pleistocene sand and gravel deposits of the Hockanum River aquifer. Materials in the aquifer range in size from clay in some western parts of town to gravel in central sections and were deposited as deltaic sequences in a series of glacial lakes by meltwater from the retreating Wisconsin ice sheet. As the glacial ice retreated northwestward, a series of deltas were formed in lakes at different water-surface altitudes. These deposits are morphosequences, similar to those described by Koteff and Pessl (1981). The lateral boundaries of the aquifer include areas of glacial till and thin, till-covered bedrock. The stratified drift and till deposits are underlain by arkosic sedimentary bedrock of the Portland Formation (Jurassic). The distribution of stratified drift, till, and alluvium and the generalized configuration and altitude of the water table is shown in figure 4.

Effective recharge to the stratified drift is much larger than to the till deposits. Estimates of long-term effective recharge in Connecticut range from 19 to 24 in/yr for stratified drift and about 9 in/yr for till deposits (Melvin and Bingham, 1991).



**Figure 4.** Surficial geology and generalized configuration and altitude of the water table in Manchester, Connecticut.



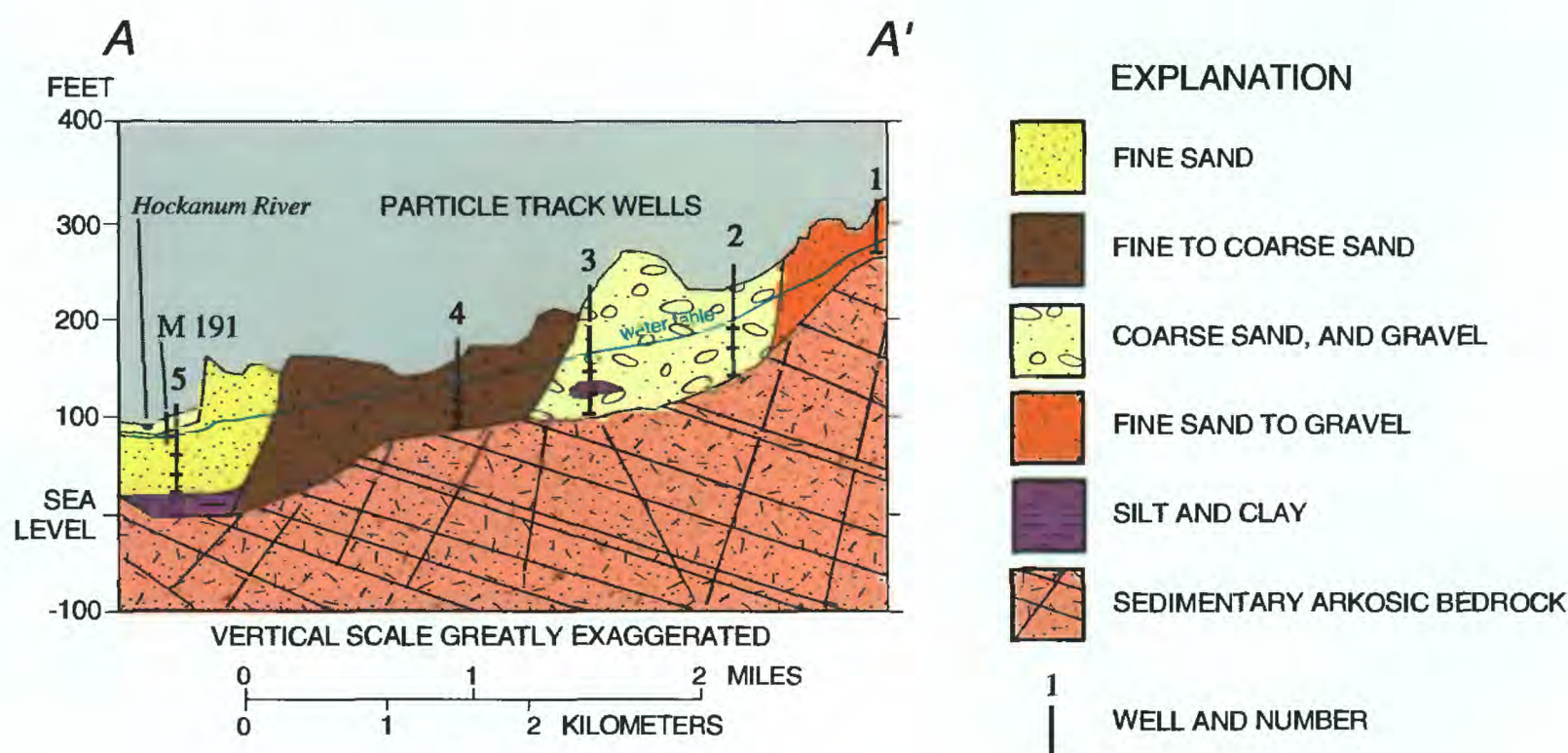
Hydraulic conductivities in the Manchester part of the aquifer range from less than 1 ft/d where fine-grained materials are dominant to about 200 ft/d where very coarse-grained materials are dominant. Transmissivities range from less than 100 ft<sup>2</sup>/d to as much as 23,000 ft<sup>2</sup>/d (Ryder and others, 1981). The aquifer is unconfined, and depth to the water table ranges from 0 (near surface-water bodies) to more than 50 ft below land surface in areas with greater topographic relief. A hydrogeologic section along the direction of ground-water flow showing the general texture of the aquifer materials is shown in figure 5.

## Water-Table Configuration and Direction of Ground-Water Flow

The water-table altitude in the study area ranges from about 300 ft in eastern upgradient sections to about 80 ft near the Hockanum River in the western sections (fig. 4). Horizontal hydraulic gradients typically range from 0.01 to 0.02 ft/ft and indicate a westward flow of ground water towards discharge points along the Hockanum River or Bigelow Brook. Vertical hydraulic gradients at ground-water-sampling locations were mostly too small to be measured accurately, with

two exceptions. At Manchester wells M 186-188 and M 179-181 (sites 3 and 4 in figs. 1 and 4), downward vertical gradients were present between the shallow and the deep wells. The downward vertical gradients averaged about 0.02 ft/ft from wells M181 to M179 (fig. 1, site 4), and 0.06 ft/ft from wells M186 to M188 (fig. 1, site 3). The downward gradients probably are related to the presence of a fine-grained layer between the intermediate and deep wells at both locations.

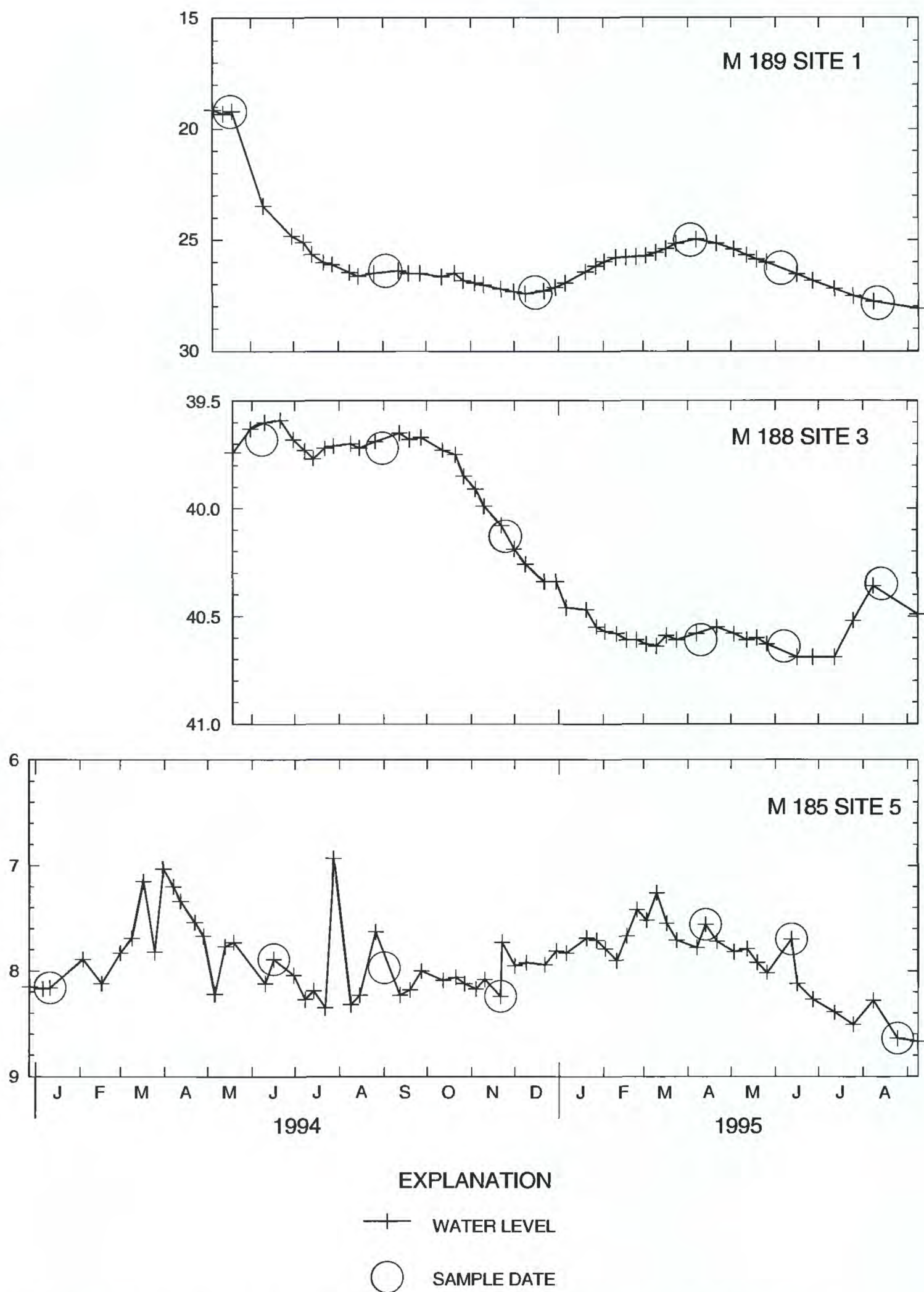
Water levels were measured weekly at flowpath wells and monthly at selected existing wells to determine the magnitude of seasonal water-table fluctuations. From January 1994 to September 1995, water-level fluctuations ranged from 1.10 ft in the coarse-grained, thick, central parts of the aquifer (site 3, well M 188) to 8.96 ft near the upgradient aquifer boundary (site 1, well M 189) (fig. 6). Water-level fluctuations were larger in wells drilled in fine-grained deposits than in wells drilled in coarse-grained deposits because of the low specific yield of the fine-grained materials. Fluctuations also were large in areas near aquifer boundaries. Water levels rose in response to periods of ground-water recharge but the timing of the recharge events varied at different locations. In the western half of the area, water levels were highest during March 1994 and 1995. In central and eastern sections of the area, water levels peaked from May to June 1994 and from April to August 1995.



**Figure 5.** General texture of aquifer materials along the direction of ground-water flow, Manchester, Connecticut. (Line of section A-A' shown in figure 1.)



WATER LEVEL, IN FEET BELOW LAND SURFACE



**Figure 6.** Seasonal ground-water-level fluctuations at selected shallow flowpath wells in Manchester, Connecticut, 1994-95.



Areal differences in ground-water-level fluctuations may be caused by a number of factors in the Manchester area, including differences in topography, unsaturated-zone thickness, and permeability; evapotranspiration; variations in rate of ground-water discharge from one section of the aquifer to another or to surface water; ground-water pumpage; and percentage of each upgradient area with impervious surfaces. Near the Hockanum River, water levels were affected by river stage (fig. 6, M 185, site 5). Water samples were collected during different water-level conditions (fig. 6) to determine if there were any seasonal variations in water quality.

## Ground-Water Discharge to Surface Water

In Manchester, ground-water and surface-water systems are hydraulically connected, and the rate of low-flow discharge at some streams in the area is representative of the rate of ground-water discharge. Low-flow measurements for discharge were made at

seven locations on three different streams draining the aquifer (fig. 1; table 4). Low-flow discharge measurements were made at selected locations on three occasions (figs. 1 and 7). Information from the low-flow measurements was used to estimate the effective recharge to the aquifer (assuming no net change in ground-water storage) and for comparison with simulated streamflow in the ground-water-flow model. In addition, the information was used to compute instantaneous loads of water-quality constituents when surface-water samples also were collected. Discharge measurements generally have an error of  $\pm 5$  percent; therefore, any load calculations made with these numbers also are subject to the same accuracy. Discharge in sections of the Hockanum River and Hop Brook may be affected by ground-water pumpage, and flows in Hop Brook also are affected by upstream surface-water withdrawals from public-supply reservoirs (fig. 1). On August 24, 1995, the streamflow gain was 3.8 ft<sup>3</sup>/s on the Hockanum River and 3.2 ft<sup>3</sup>/s at Bigelow Brook, indicating a ground-water discharge of 7 ft<sup>3</sup>/s from this section of the aquifer.

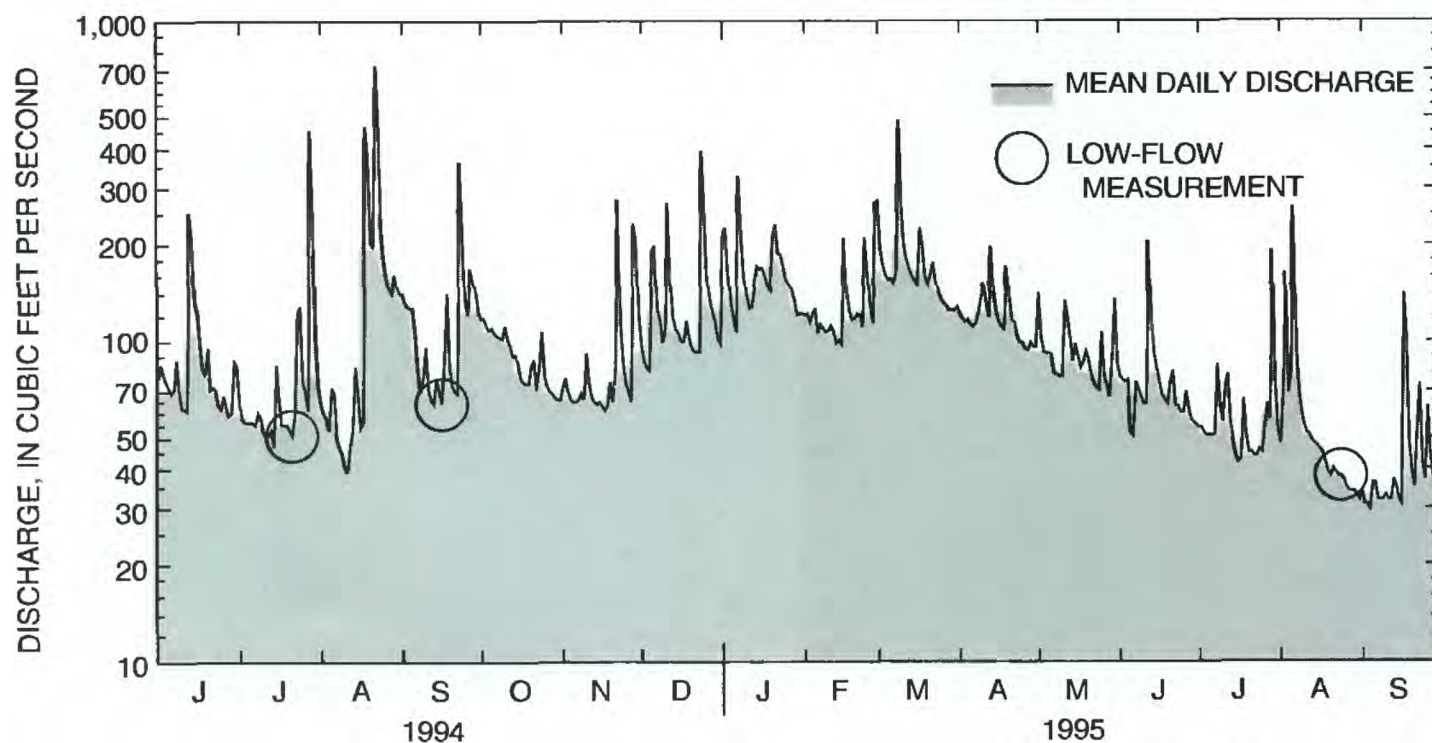
**Table 4.** Low-flow discharge measurements and comparison with ground-water-flow model output, Manchester, Connecticut

[mi<sup>2</sup>, square mile; ft<sup>3</sup>/s, cubic foot per second]

Site designation (see fig. 1)	Surface-water-measurement location	U.S. Geological Survey station No.	Drainage area (mi <sup>2</sup> )	Date	Instantaneous discharge (ft <sup>3</sup> /s)	Streamflow gain from nearest upstream station (ft <sup>3</sup> /s)	Streamflow gain simulated in MODFLOW (ft <sup>3</sup> /s)
A	Bigelow Brook above Center Springs Pond	01192321	1.4	8-24-95	0.26		
B	Bigelow Brook at Broad Street	01192322	1.8	8-24-95	.49	0.23	0.91
C	Bigelow Brook at Hilliardville	01192323	3.00	7-21-94	4.2		
				9-16-94	5.0		
				8-24-95	3.2	2.71	2.46
D	Hockanum River at Railroad Bridge	01192310	55.1	7-21-94	31.3		
				9-16-94	42.2		
				8-24-95	23.5		
E	Hockanum River, Adams Street, Manchester,	01192315	56.1	9-16-94	43.1	.9	
F	Hockanum River downstream of New State Road	01192332	59.4	7-21-94	38.8		
				9-16-94	46.0	<sup>1</sup> 3.8 1.2	
				9-16-94	48.0	<sup>1</sup> 5.8 0.8	
				8-24-95	30.5	<sup>1</sup> 7.0 3.8	3.63
H	Hop Brook near Manchester	01192480	11.7	7-21-94	2.42		

<sup>1</sup>Measured flow of Bigelow Brook has been subtracted for comparison with MODFLOW simulation.





**Figure 7.** Discharge of the Hockanum River near Manchester (site G, fig. 1) and timing of low-flow measurements in Manchester, Connecticut, 1994-95.

## Simulation of Ground-Water Flow

A two-dimensional, steady-state ground-water-flow model was used to better understand the ground-water-flow system and to help estimate the travel distance and location of source areas for ground water flowing through the sampled wells. Knowing the land use in the predicted source area allows for a better interpretation of the water-quality data and the differences in water quality that occur between different depths in the aquifer. Ground-water flow was simulated using the finite-difference model (MODFLOW) developed by McDonald and Harbaugh (1988). A particle-tracking analysis to determine flowpaths to the sampled wells was applied using MODPATH (Pollock, 1994).

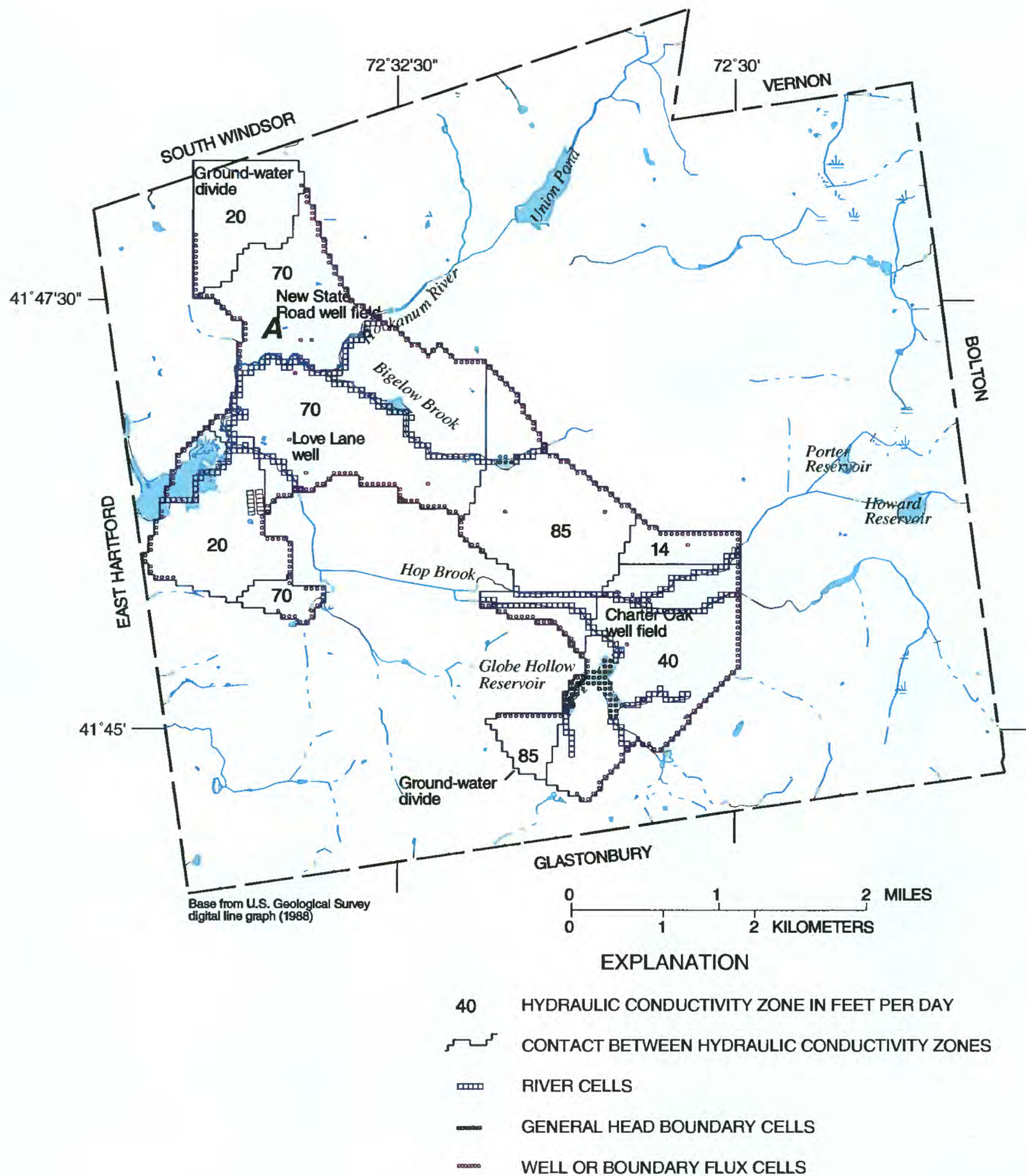
### Description of Flow Model and Model Assumptions

The model contains a finite-difference grid with 108 rows and 116 columns; columns are aligned north-south. Individual cells are square and are 200 by 200 ft. The location of the model area is shown in figure 8. The model is a 2-dimensional flow model, which has some limitations that may affect the results of the simulation when compared with actual conditions. All flow is assumed to be horizontal through an isotropic

medium. The two-dimensional model is not able to simulate any vertical-flow components or any vertical variations in hydraulic conductivity. Head differences measured among clustered wells in this study were minimal; therefore, the simulation is expected to give reasonable results. However, because the presence of confining or semi-confining layers is not represented, in some instances flowpaths may be longer or shorter than those predicted by the model. Differences between actual and simulated flowpaths are not expected to significantly affect the results of this study.

The ground-water-flow model was used to simulate steady-state conditions. This means that the simulated water-table altitude and aquifer stresses are assumed to represent long-term, average conditions. Water-level data at USGS long-term observation well SW64 in a stratified-drift aquifer in nearby South Windsor were analyzed to determine if conditions in Manchester during the study were similar to long-term average conditions. Water-level statistics for well SW64 were computed for 1966-95 and compared with water-level statistics from October 1993 to September 1995. The normal (middle 50 percent of the data) range in water level for well SW64 was 10.13 to 12.03 ft below land surface. The mean water level in well SW64 from October 1993 to September 1995 was 10.27 ft below land surface.





**Figure 8.** Location of model boundaries and simulated hydraulic conductivity zones, Manchester, Connecticut.



This indicates that water levels in a nearby aquifer were slightly higher during the study period but within the normal range. Ground-water levels in most of the study area did not change by much more than 2 ft during the study, indicating that ground-water conditions did not change substantially (fig. 3). Exceptions were in areas near the aquifer boundaries or near public-supply wells.

### Boundary Conditions

The model boundaries generally reflect the physical boundaries of the stratified-drift aquifer in Manchester. The lateral boundaries, with several exceptions, were constructed to coincide with the 10-foot saturated-thickness contour in the stratified-drift aquifer. In most areas, this boundary is just inside the contact between the stratified drift and the till or till-covered bedrock. Because a large amount of water may enter the model area as recharge from upland areas outside the model boundaries, but within the surface water basin, a constant-flux boundary was used and simulated as a series of recharging wells. Fluxes were calculated by measuring the area of stratified drift and till in segments that are adjacent to the model but within the surface-water drainage divide. The fluxes were estimated using effective recharge rates of 20 in/yr to stratified-drift deposits and 8 in/yr to till deposits. These fluxes were apportioned among model cells along each boundary segment.

Other lateral boundaries in the model are ground-water divides in the northwestern and southern parts of the town. These ground-water divides are present in bedrock valleys with relatively thick stratified drift. The boundary in the southwestern corner of the model area is a no-flow boundary and a geologic boundary, because the model boundary is perpendicular to water-table contours and is near the contact between coarse-grained stratified drift and fine-grained silt and clay deposits. The lower boundary of the model is the contact between the stratified drift and the underlying till or bedrock. For the purposes of this simulation, the underlying till and bedrock are considered to be impermeable. The upper boundary is the water table.

Other internal boundaries in the model are rivers and ponds. The streambeds and pond bottoms were assumed to be leaky confining beds and were simulated as head-dependent flux boundaries. Streams were simulated using the River package in MODFLOW. Streambed conductance was defined for each river cell in the model by the equation:

$$C_s = \frac{KLW}{M}, \quad (2)$$

where

- $C_s$  is the streambed or pond-bottom hydraulic conductance,
- $K$  is the vertical hydraulic conductivity of the streambed or pond bottom,
- $L$  is the length of the stream reach within a given node,
- $W$  is the width of the streambed within a given node, and
- $M$  is the thickness of the streambed.

Ponds were simulated using the General Head Boundary package in MODFLOW. Pond-bottom conductance was determined using the same method as for streams, except that  $L$  and  $W$  are the length and width of nodes within the area of the pond. There are other important differences between the River package and the General Head Boundary package in MODFLOW. In the River package, flow out of the stream is limited by a maximum head difference between the stream and the aquifer that is equal to the stream stage minus the bottom of the streambed elevation. Flow into the stream is not limited and is governed by the head difference between the aquifer and the river and the conductance. In the General Head Boundary package, there is no such limitation, and flow between the pond and the aquifer is governed by the head difference and the conductance, regardless of the direction of the flow. In this simulation, all rivers and ponds were assumed to be 3 ft deep, with a streambed or pond-bottom thickness of 3 ft. Vertical hydraulic conductivity of streambed sediments was set to 4 to 20 ft/d, and vertical hydraulic conductivity of pond-bottom sediments was set to 0.1 ft/d. These values were based on a knowledge of riverbed and pond-bottom sediment composition and are similar to values used in other studies in New England (de Lima, 1991).



## Aquifer Properties

Horizontal hydraulic conductivities were assigned to zones in the aquifer on the basis of a geologic model. The stratified-drift deposits in Manchester consist of a series of overlapping glacial deltaic deposits that represent several ice-marginal positions. These sequences of deposits in Manchester were used as model zones where uniform hydraulic conductivities were assigned. The zones were defined using 1:24,000-scale maps being used to construct a map of Quaternary deposits in Connecticut (J.R. Stone, U.S. Geological Survey, written commun., 1996). Textural changes may be present in each sequence, primarily the fining of materials with depth and distance from the source area of glacial meltwater. For the purpose of this investigation, however, average horizontal hydraulic conductivities that represent the range of materials were assigned to each model zone. Initial estimates of hydraulic conductivity used in the model were based on grain-size analyses of stratified-drift samples obtained for this investigation. The method uses a median particle size and is described in Mazzaferro and others (1979). The hydraulic conductivities assigned to each zone and used in the final simulation are shown in figure 8. Some deposits were combined into a single zone if the hydraulic conductivities were similar. Horizontal hydraulic conductivities ranged from 14 ft/d for fine-grained deposits, including fine sand, silt and clay, to 85 ft/d for coarse to very coarse sand with some gravel. A value for porosity is required in the particle-tracking simulations because linear velocity must be calculated. A porosity of 0.4 has been used in particle-tracking simulations in this report. This value is similar to those used for simulations on Cape Cod (Masterson and others, 1996), in which 0.35 was assigned to glacial outwash deposits. Porosity also was partly determined by calibrating the particle tracking in MODPATH using the age-determination results. Porosity values were adjusted in the simulations until traveltimes for particles most closely matched the  $^3\text{H}$ - $^3\text{He}$  dates.

## Stresses to the Model

Stresses to the aquifer system that were simulated by the model include ground-water recharge and pumping at public-supply wells. Because the model was simulated as steady state, the stresses applied to the aquifer should represent long-term,

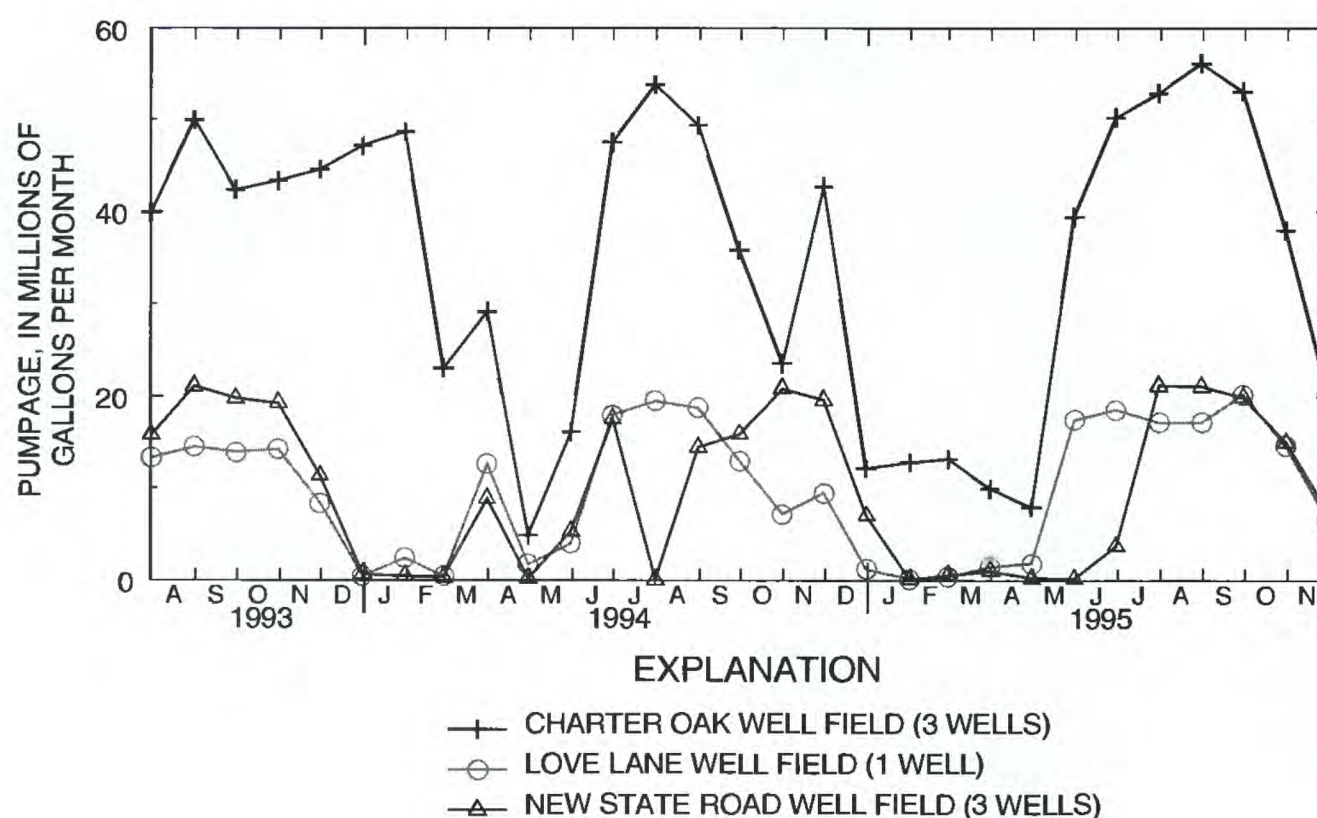
average conditions. A value of 20 in/yr was used in the simulation for ground-water recharge; this value has been commonly used in Connecticut for recharge in areas of coarse-grained stratified drift (Melvin and Bingham, 1991). If the low-flow data in table 4 are extrapolated for an entire year, streamflow gains in the stratified drift would range from 12.7 to 31 in/yr; these values bracket the estimated recharge rate. The recharge value used in the simulation is an effective recharge rate and includes the effects of evapotranspiration. All recharge is assumed to come from precipitation, but in an urban area such as Manchester, other sources of recharge may include leaky water mains, sanitary sewer lines, and lawn irrigation.

Pumping was simulated from three well fields with seven public-supply wells in the stratified-drift aquifer. Average pumping rates were assigned on the basis of data obtained from the Manchester Water Department for July 1993 to November 1995, which generally corresponded to the time when ground-water samples were collected and water levels were measured (October 1993 to October 1995). Several public supply wells are metered collectively; in such cases, pumpage was divided evenly among the wells. Ground-water pumpage from the three well fields for 1993-95 are shown in figure 9.

## Model Calibration and Hydrologic Budget

The model was calibrated by comparing computed heads with average water levels (from 1993 to 1995) at 10 observation wells. Values of hydraulic conductivity were then slightly adjusted in zones where the heads did not match well. Other parameters in the model, including recharge, river stage, streambed, and general-head boundary conductance, also were changed through a series of multipliers to get the best fit. The mean root squared error was used as the basis for the calibration. Because the altitude of the well locations was measured from the USGS quadrangle map of Manchester (1:24,000) to an accuracy of about 5 ft, the calibration was considered acceptable if all simulated heads were within 5 ft of the measured heads. (table 5). In addition, discharge in sections of three streams was compared with low-flow measurements to ensure that similar flows were simulated (table 4). The hydrologic budget calculated from the ground-water-flow model is shown in table 6.





**Figure 9.** Pumpage from well fields in the stratified-drift aquifer in Manchester, Connecticut, July 1993 to November 1995. (Locations of well fields shown in figure 1.)

**Table 5.** Comparison of average measured and steady-state simulated heads, Manchester, Connecticut, 1993-95

[ft, foot]

Local well No. (fig. 1)	Average measured head (ft above sea level; (10/93-9/95)	Steady-state simulated head (ft)	Head difference (ft)
M 158	215.18	213.95	-1.23
M 165	151.61	149.81	-1.80
M 168	87.04	85.66	-1.38
M 171	222.51	224.71	2.20
M 172	282.61	277.79	-4.82
M 178 (site 2)	193.67	191.51	-2.16
M 181 (site 4)	126.37	130.83	4.46
M 185 (site 5)	83.11	79.46	-3.65
M 188 (site 3)	159.81	163.00	3.19
M 189 (site 1)	274.16	272.69	-1.47
Mean root squared error .....			2.91

**Table 6.** Model-calculated hydrologic budget for steady-state conditions

Budget Item	Volumetric rate	
	million gallons per day	cubic feet per second
<b>Inflow</b>		
Effective recharge from precipitation.....	6.36	9.85
Flow into model from adjacent areas.....	4.82	7.46
Leakage from streams.....	2.86	4.42
Leakage from ponds .....	.57	.88
Total inflow.....	14.6	22.6
<b>Outflow</b>		
Discharge from pumping wells.....	1.88	2.91
Discharge to rivers.....	12.00	18.57
Discharge to ponds .....	.73	1.13
Total outflow.....	14.6	22.6

## Conceptualized Flowpaths to Sampled Wells

Ground-water ages were determined for 10 samples from flowpath wells screened from less than 5 ft to nearly 58 ft below the water table. At three locations where  $^3\text{H}$ - $^3\text{He}$  samples were not collected,

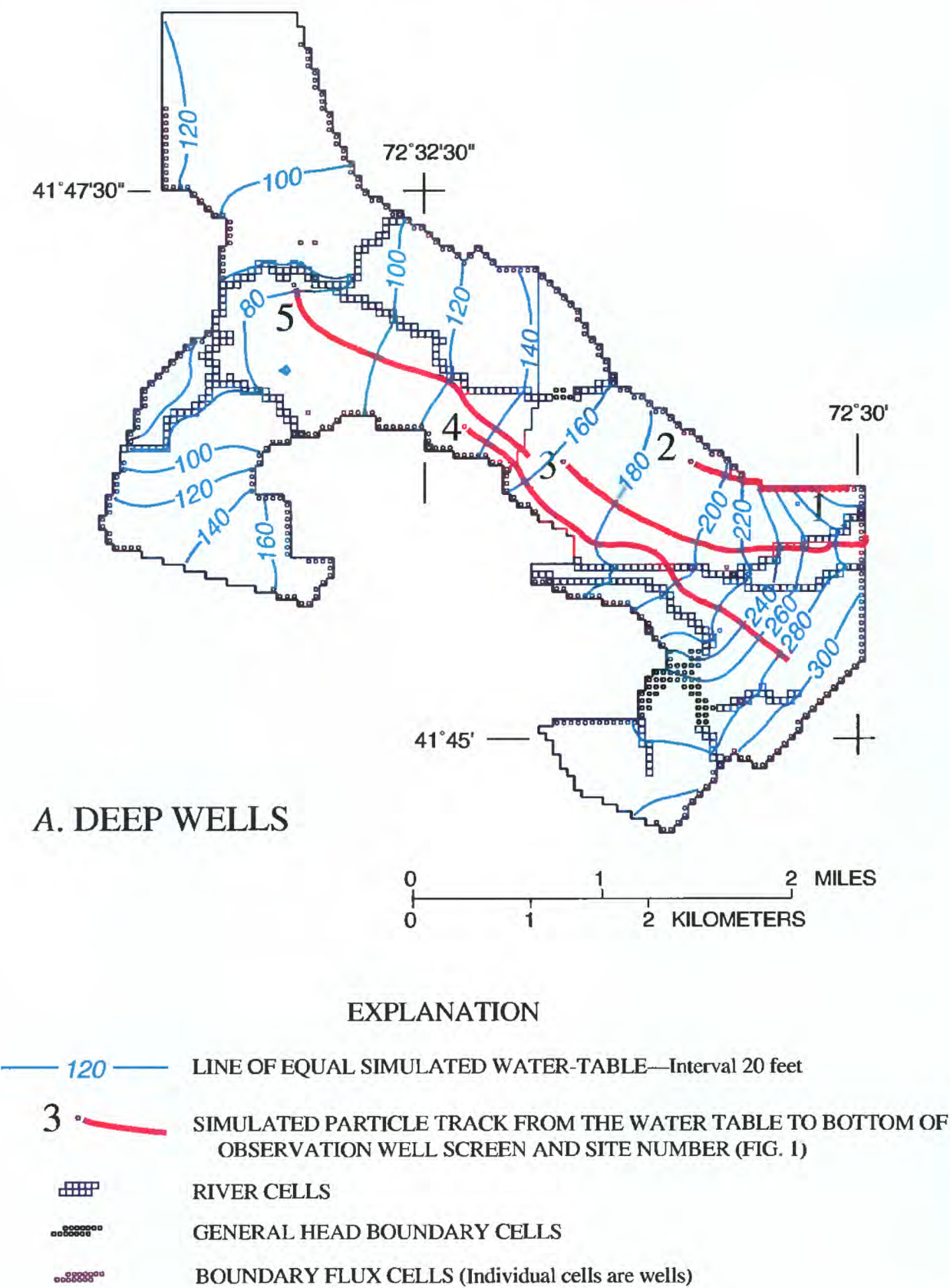
ages were estimated on the basis of results of the MODFLOW/MODPATH simulation. The ground-water samples range in age from 0.9 to 22.6 years. This indicates that samples from different locations and depths represent recharge from different time periods, when land use or hydrologic conditions may have been different. The age of the ground water, the method of



determination, well depth, depth of sample below water table, and simulated travel distance from source area are shown in table 7.

The age of the ground water in Manchester was used to calibrate the particle-tracking algorithm in MODPATH. Particle tracking was used to simulate the flowpaths to the sampled wells to determine the travel distance from the source area and its location (table 7). Particles were added to the model at sampled well locations at depths coincident with the top and bottom of the

screened interval and were then moved backward in time. This was done to estimate differences in travel distance from the source areas to the top and bottom of simulated 5-foot observation well screens. In some cases, the determination of travel distance was limited because the particle tracks to the wells began in areas outside the model boundaries. Model outputs show that the wells lie along several ground-water flowpaths. The simulated particle tracks from the source areas to the sampled wells and simulated water-table altitude are shown in figure 10.



**Figure 10.** Simulated water-table altitude and particle tracks to the bottom of the well screen at (A) deep, (B) intermediate depth, and (C) shallow wells in Manchester, Connecticut.



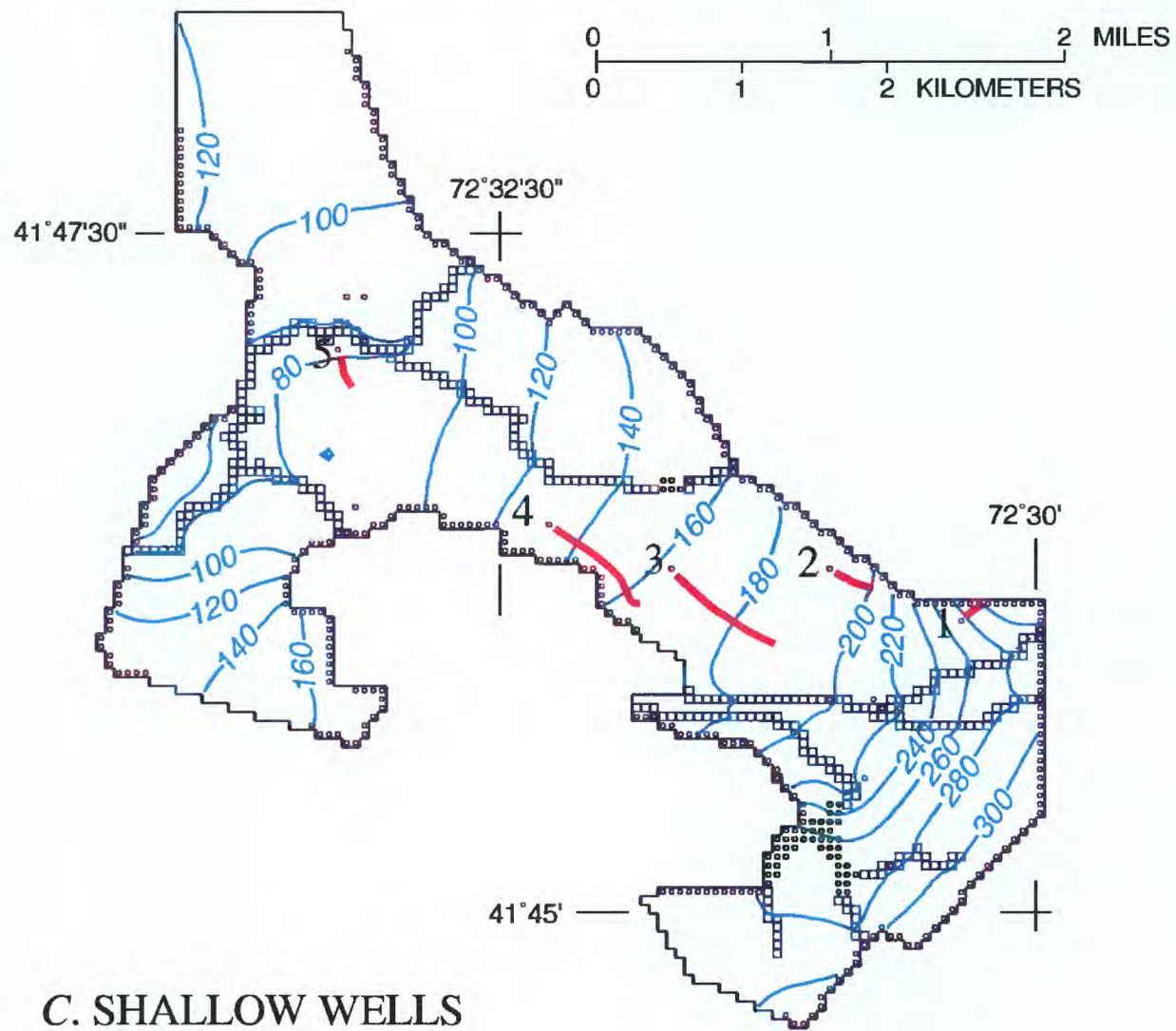
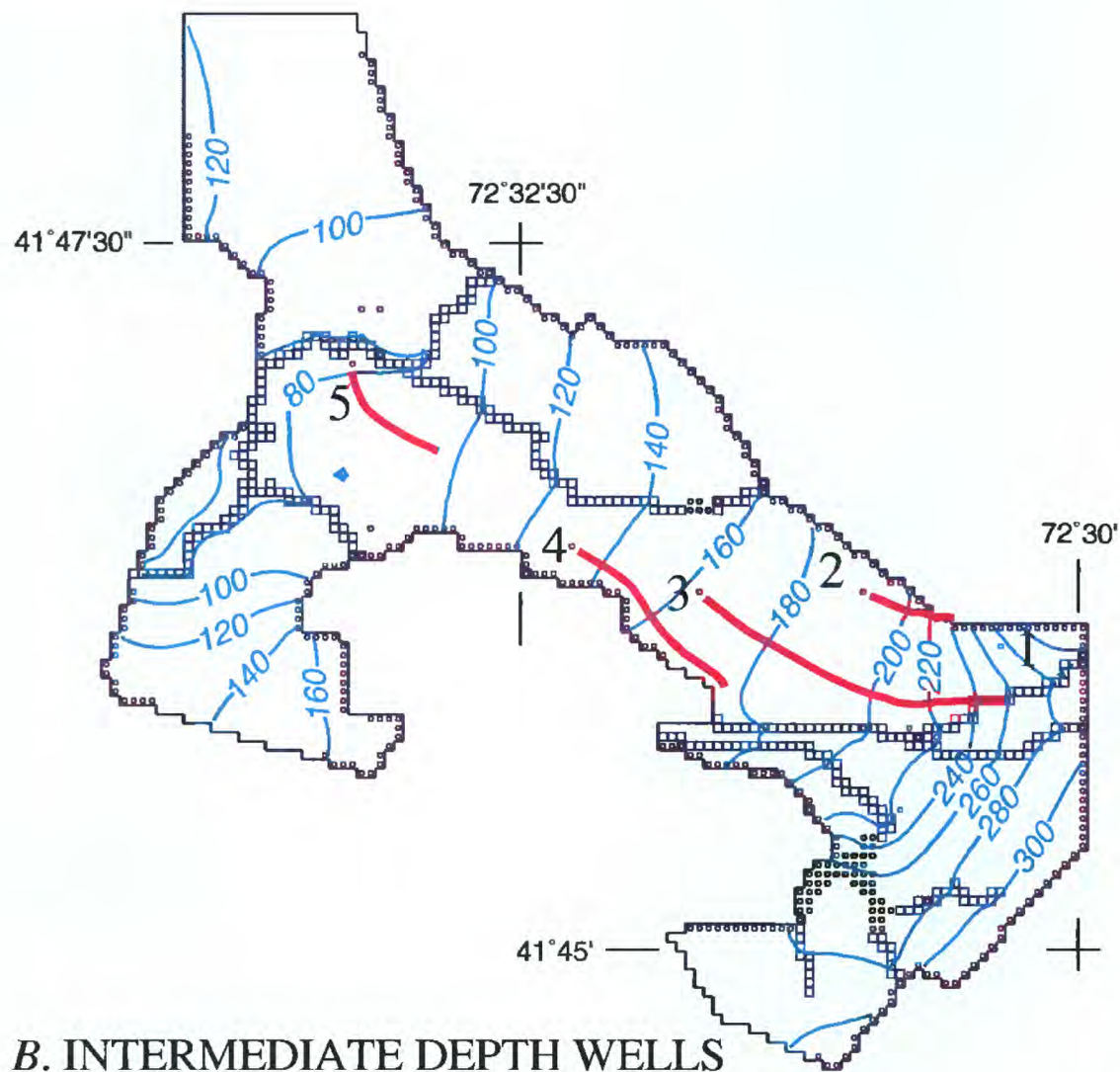


Figure 10.—Continued



**Table 7.** Age of ground water and simulated travel distance from source areas at selected wells in Manchester, Connecticut, June 1995

[ft, foot; >, actual value is greater than value shown]

Site (fig. 1)	Local well No.	Age (years)	Depth of well (ft below land surface)	Depth to bottom of sampling interval (ft below water table)	Simulated travel distance from source area (ft)	
					Maximum	Minimum
1	M 189	<sup>1</sup> 1	31.0	4.7	600	50
2	M 176	5.5	90.0	56.5	>2,000	>2,000
	M 177	1.9	64.4	31.0	>2,000	2,000
	M 178	.9	44.3	10.7	1,000	300
3	M 186	<sup>1</sup> >14	100	57.8	>9,000	>9,000
	M 187	10.5	76.6	36.1	7,000	6,000
	M 188	.7	54.4	13.8	3,000	2,200
4	M 179	19.6	64.7	44.7	11,000	7,000
	M 180	<sup>1</sup> 6	45	25.1	5,000	4,150
	M 181	1.9	31.4	11.3	2,500	2,000
5	M 183	22.4	60.3	51.8	8,000	5,000
	M 184	22.6	42.8	33.3	3,000	2,150
	M 185	11	23.7	16.0	1,000	700

<sup>1</sup>Estimated from MODFLOW with MODPATH particle tracking.

**Table 8.** Land use/land cover and population density in simulated source areas to flowpath wells, Manchester, Connecticut

[Land use/land cover classification is modified from Civco and Hurd (1990). No., number. An x is placed in the box if the recharge area contained a particular category]

Site (fig. 1)	Local well No.	Population density, 1990 (people per mi <sup>2</sup> )	Land use/land cover classification				
			High- density residential and commercial	Medium- density residential	Forest	Turf or Agricul- tural	Major road
1	M 189	3,400		x		x	
2	M 176	6,544	x	x	x	x	x
	M 177	6,544	x	x	x	x	x
	M 178	8,563	x	x			
3	M 186	629		x	x		x
	M 187	1,346	x	x		x	x
	M 188	5,160	x	x	x	x	
4	M 179	940	x	x	x	x	x
	M 180	8,808	x	x	x		
	M 181	8,895	x	x			
5	M 183	3,952	x	x	x	x	x
	M 184	5,923	x	x			x
	M 185	3,400	x			x	x

Land use was determined qualitatively because there is some uncertainty in the location of the source area for each well, and some of the particle tracks begin at or beyond the model boundaries. Land use (Civco and Hurd, 1990) was examined in the simulated source areas for model particles that passed through the top and the bottom of the well screen at each location. The land use was then inspected along the direction of ground-water flow between the two surface locations, as well as in a 500-foot buffer zone on either side of this line.

Analysis of land-use data shows that source areas to all flowpath wells contain at least some medium- to high-density residential and commercial areas. These areas contain numerous streets, single and multi-family dwellings with lawns, and some commercial buildings. Land use in the source areas to the flowpath wells, can, with minor exceptions, be considered urban. In addition, population density of the recharge area for each well was determined from the 1990 census block group data (U.S. Department of Commerce, 1991), and it ranged from 629 to 8,895 people/mi<sup>2</sup>. Land uses in the source area for each well and population density are shown in table 8.



## WATER QUALITY IN THE SURFICIAL AQUIFER

During 1994-95, water samples were collected at 14 wells and analyzed as many as 7 times for inorganic constituents (major ions, nutrients, and field parameters), once for trace elements, twice for pesticides, and four times for VOCs (fig. 3). Samples were compared, where possible, by depth in the aquifer, age of the ground water, and the time of year the samples were collected. Analysis of land use and population data was used to help understand possible sources of selected constituents. Surface-water-quality data collected during periods of low streamflow also were analyzed to determine the effects of ground-water discharge from the Manchester area on surface-water quality in the Hockanum River and other tributaries.

### Major Ions

Water type in the Manchester area ranges from calcium bicarbonate to sodium chloride (fig. 11). Calcium and sodium are the dominant cations, and bicarbonate and chloride are the dominant anions. Each of these ions typically comprises at least 20 percent of the total dissolved solids. Magnesium is a minor percentage of the cations, usually around 10 percent or less, and sulfate and nitrate combined usually amount to 13 to 25 percent of the total anions. No concentrations of major ions or nutrients exceeded U.S. Environmental Protection Agency (USEPA) maximum contaminant levels (MCLs) (U.S. Environmental Protection Agency, 1996).

Concentrations of selected major ions and values of field parameters were measured in ground-water samples from five shallow flowpath wells screened at the water table. These were compared with concentrations of 10 samples collected in undeveloped locations from water-table wells in aquifers underlain by stratified drift derived from arkosic bedrock in central Connecticut (table 9). The 10 wells were installed and sampled as part of the NAWQA land-use study to determine constituent concentrations at ambient conditions.

Median concentrations of several constituents associated with human activities, including sodium, chloride, and nitrite plus nitrate as nitrogen, were

**Table 9.** Comparison of median values of selected field parameters and concentrations of constituents in water samples from 5 shallow flowpath study wells in Manchester and 10 water-table wells in undeveloped areas in Connecticut with similar aquifer composition

[Samples from shallow flowpath study wells: Medians calculated by using median value for multiple samples from individual wells. NAWQA samples: Collected in undeveloped areas underlain by stratified-drift aquifers derived from arkosic bedrock]

Constituent or property	Samples from shallow flowpath study wells (median value)	NAWQA samples (median value)
Specific conductance, field ( $\mu\text{S}/\text{cm}$ ).....	400	170
pH, field (standard units).....	6.4	6.9
Alkalinity, field (mg/L).....	94	31
Bicarbonate, field (mg/L as $\text{CaCO}_3$ ).....	120	37
Hardness (mg/L as $\text{HCO}_3$ ).....	130	63
Noncarbonate hardness (mg/L)	40	11
Calcium (mg/L).....	45	18.5
Magnesium (mg/L).....	5	4
Sodium (mg/L).....	23	6
Potassium (mg/L).....	1.1	1.2
Sulfate (mg/L).....	19	13
Chloride (mg/L).....	46	7.9
Bromide (mg/L).....	.05	.02
Silica (mg/L).....	15	15
Dissolved solids, residue at 180°C (mg/L).....	230	100
Nitrite plus nitrate as nitrogen (mg/L).....	3.00	.4

higher in samples from the Manchester wells than in samples from wells in the undeveloped locations. Sodium and chloride concentrations in samples from all wells in the Manchester area were higher than the median value for the wells in the undeveloped locations (table 9). The most likely source of elevated chloride concentrations is runoff of deicing chemicals applied to major and secondary roads, sidewalks, parking lots, and driveways. Because concentrations of sodium and chloride correlate well, these compounds were likely applied primarily as sodium chloride salt. The simulated source areas for all flowpath wells contain paved surfaces where road salt probably was applied. Differences in sodium and chloride concentrations

## EXPLANATION

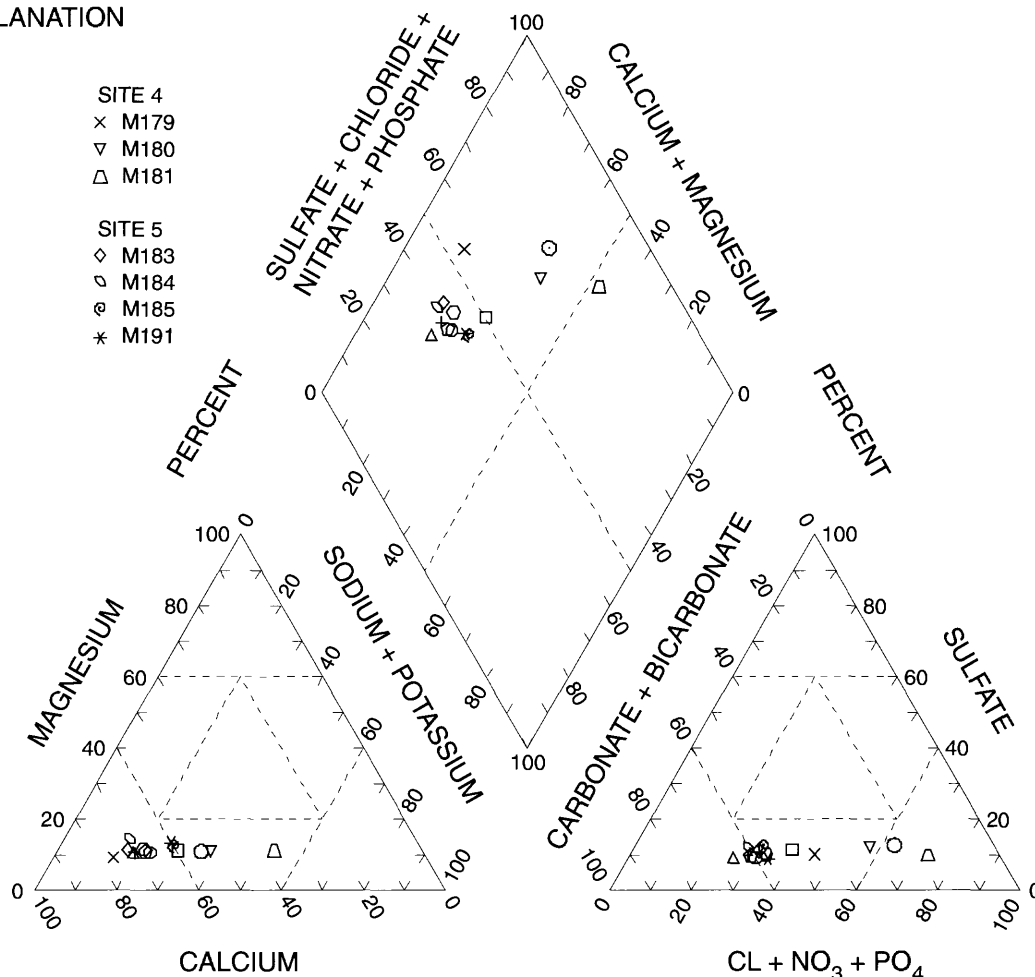
SITE 1  
○ M189

SITE 2  
+ M176  
△ M177  
□ M178

SITE 3  
◇ M186  
○ M187  
○ M188

SITE 4  
× M179  
▽ M180  
△ M181

SITE 5  
◇ M183  
▽ M184  
⊙ M185  
\* M191



PERCENTAGE OF TOTAL MILLIEQUIVALENTS PER LITER

**Figure 11.** Chemical composition of water samples collected from wells in the Hockanum River aquifer in Manchester, Connecticut, August 1995. [CL; chloride,  $\text{NO}_3$ ; nitrate,  $\text{PO}_4$ ; phosphate]

among flowpath wells may relate to other factors, such as the amount of paved surfaces in the source area and the age of the water. Another possible source of sodium and chloride is exfiltration from leaky sanitary and storm sewers.

Median concentrations of major ions associated with both natural and anthropogenic sources, including calcium, bicarbonate, and sulfate, also were higher in samples from the wells in Manchester than in samples from wells in the undeveloped location. Grady (1994, p. B32) indicates that calcium concentrations were significantly higher in urban areas than in undeveloped areas studied in Connecticut. Also, the median

bicarbonate concentration in the shallow flowpath wells (120 mg/L) was much higher than the median concentration in undeveloped areas (37 mg/L). The percentage of these compounds that may be derived from human activities is difficult to distinguish, but possible sources of the high concentrations of these compounds include calcium carbonate (lime) applications to lawns and gardens and laundry detergents from leaky sanitary sewer lines. The presence of high concentrations of calcium and bicarbonate are responsible, in part, for elevated values of specific conductance, alkalinity, hardness, and dissolved solids.

Comparisons were made to determine if the concentrations of inorganic constituents differed with depth in the aquifer. Wells were grouped by depth of screen or completion level below the water table (shallow, intermediate, and deep) and concentrations of selected constituents in wells screened at different depths at the same site also were compared to make this determination. Most constituents and properties generally did not differ consistently with depth, but at individual sites, many constituent concentrations did differ with depth. For example, concentrations of sodium and chloride generally were higher in samples from shallow wells than from intermediate or deep wells. Chloride concentrations plotted as a function of depth of sample below water table are shown in figure 12. Chloride concentrations with depth and time at site 4 are shown in figure 13.

The lack of a consistent relation between depth of sample below the water table and concentrations of inorganic constituents indicates that other factors may

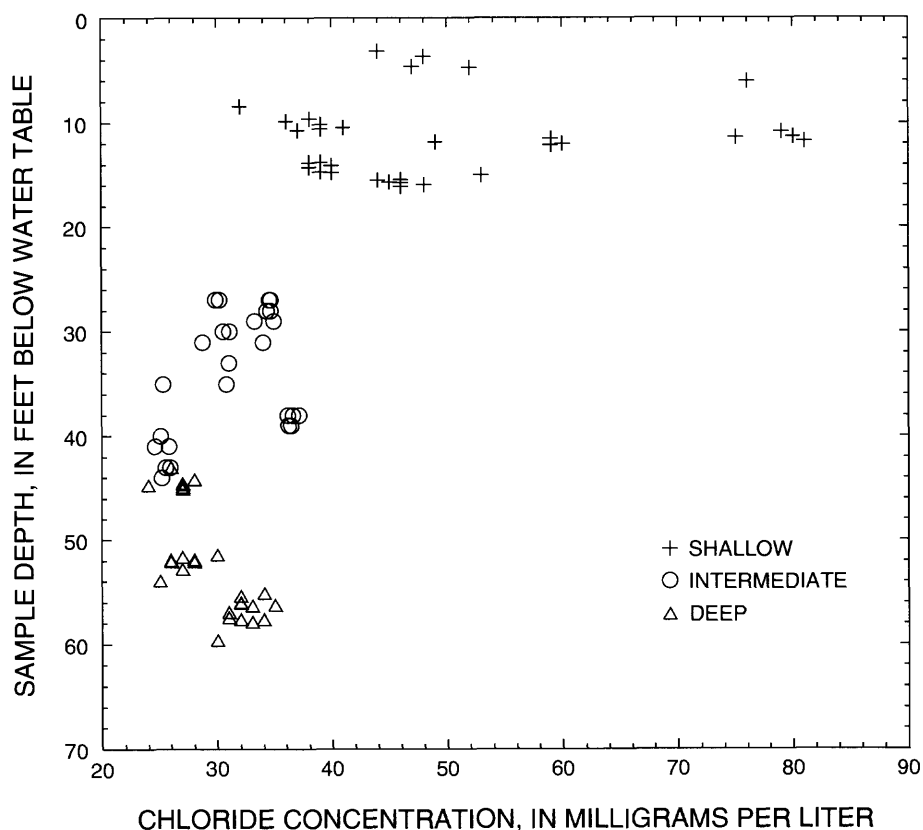
affect concentration, including flowpaths and travel times to individual wells, land use in the source area, and interaction with different geologic materials.

Concentrations of inorganic constituents were then compared by age of ground water. Samples collected in Manchester were classified into two groups—old (10 to 22.6 years) and young (0.9 to less than 10 years). The ages are based primarily on the results of the  $^3\text{H}$ - $^3\text{He}$  analysis of 10 samples and age calculations from the ground-water-flow model at locations where  $^3\text{H}$ - $^3\text{He}$  analysis was not done.

Median values of dissolved oxygen, pH, bicarbonate, alkalinity, total hardness, and calcium differed by age of water (table 10). These differences may relate to the length of residence time in the aquifer and the interaction with geologic materials. Median concentrations of sodium, chloride, and nitrite plus nitrate nitrogen were higher in samples of young water than in old water. These constituents are more likely to have anthropogenic sources for high concentrations

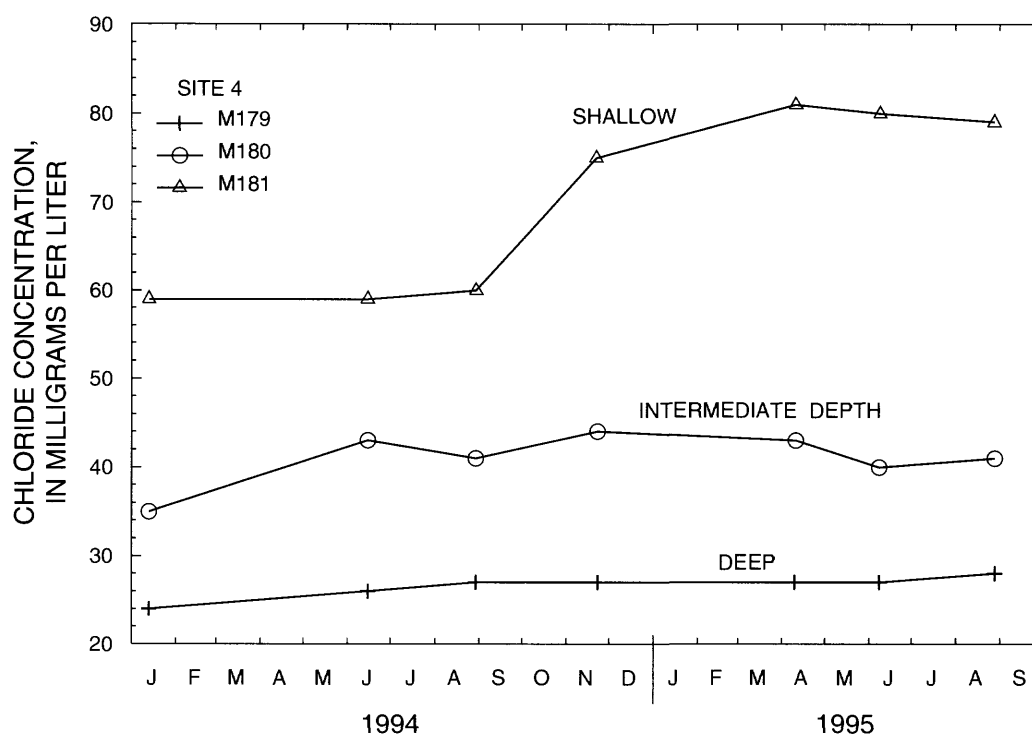
than some other constituents. Higher sodium, chloride, and nitrate concentrations in the younger samples than in the older samples may indicate an increase in application rate of these constituents with time or changes in the land use in the source area to the wells over time. For example, the high chloride concentrations may indicate an increase in the use of road salt during the years the younger waters were recharged or could indicate a source area with more paved surfaces and a greater overall use of deicing chemicals in winter.

Water samples were collected as many as seven times at each well to help determine the temporal variability of inorganic water-quality constituents. Sample times were chosen to represent different or varying hydrologic conditions that occurred during the study period (January 1994



**Figure 12.** Chloride concentrations as a function of depth of sample below the water table, Manchester, Connecticut, 1994-95.





**Figure 13.** Chloride concentrations in samples from wells at site 4, Manchester, Connecticut, 1994-95.

**Table 10.** Median concentrations of selected water-quality constituents by age of ground water, Manchester, Connecticut

[Median values were used for individual wells with multiple samples; mg/L, milligram per liter]

Constituent or property	Median concentrations in samples of ground water	
	Young (0 to less than 10 years)	Old (10 to 22.6 years)
pH, field (standard units).....	7.0	7.7
Dissolved oxygen (mg/L).....	8.5	6.0
Alkalinity, field (mg/L) .....	94	110
Bicarbonate, field (mg/L as CaCO <sub>3</sub> ) .....	120	140
Hardness (mg/L as HCO <sub>3</sub> ) ....	130	150
Calcium (mg/L).....	45	53
Sodium (mg/L).....	21	16
Chloride (mg/L) .....	39	30
Nitrite plus nitrate as nitrogen (mg/L) .....	3.8	2.6

to September 1995). (See fig. 6 for the sample-time distribution in relation to the water levels at three sites in different parts of the ground-water-flow system.) Most inorganic-water-quality constituents and field parameters generally displayed some temporal variability during this period. Constituent concentrations ranged at some wells from 10 to 40 percent of the median value. The greatest fluctuations in concentration of inorganic constituents generally were in samples from wells screened at 10 ft or less below the water table. Also, concentrations of inorganic constituents varied more in samples from wells with the largest water-level fluctuations. Not enough samples were collected over a sufficient period of time to determine if seasonal or cyclical patterns might be present; however, at specific sites, some seasonal variations were observed that may be attributed to processes on the land surface.

Chloride concentrations at site 4 displayed some seasonal variability (fig. 13). The highest and most variable chloride concentrations of the three wells at this site were in the shallow well (screened about 10 ft below the water table). Chloride concentrations were less variable in the intermediate well (screened in the middle of the saturated zone) and concentrations were lowest and the least variable in the deep well (screened at the bottom of the saturated zone). Because a major source of high chloride concentrations is road-salt applications, some seasonal variability in chloride concentrations would be expected during periods of snowmelt and subsequent runoff and infiltration; however, the chloride concentration at site 4 increased from August to November 1994 (fig. 13). The timing of the recharge event that produced the high chloride concentrations is difficult to precisely determine because water from this well is about 2 years old, and the sampled water is likely an integration of more than 1 year of recharge. Point samples collected at the water table (from wells with short screens) would be needed to accurately determine temporal or seasonal differences in concentrations of constituents derived largely from surface sources. Variation was less at the deeper wells at this site, due to muting of seasonal inputs by dispersion and mixing over time, or differences in upgradient source concentrations. Fluctuations in concentrations of inorganic constituents also may result from minor changes in ground-water flowpaths that occur in response to water-level fluctuations and variations in recharge, rather than temporal differences in the input of chemicals from anthropogenic sources.

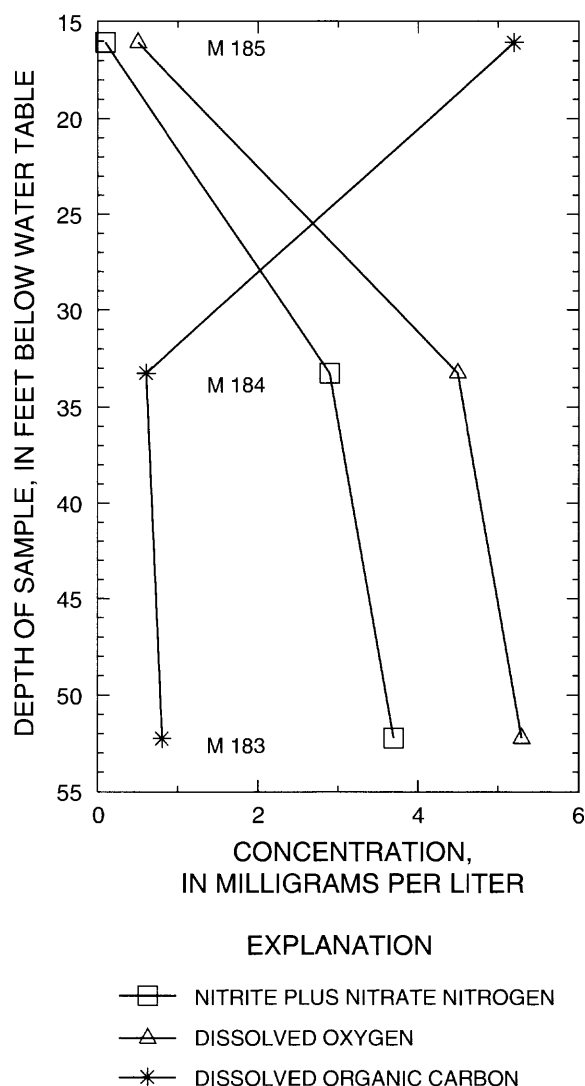
## Nitrogen

Elevated concentrations of nitrite plus nitrate nitrogen with respect to samples collected in undeveloped areas were present in 13 of the 14 flowpath wells sampled. The median concentration in samples from flowpath wells in Manchester was 3.00 mg/L, which is higher than the median concentration of 0.4 mg/L in undeveloped areas with similar aquifer composition (table 9).

Concentrations of nitrite plus nitrate nitrogen were elevated throughout most of the areal extent and thickness of the aquifer. This indicates that the effects of human activities have reached the bottom of the

surficial aquifer; however, because the age dating by  $^3\text{H}$ - $^3\text{He}$  indicated that the oldest waters may date back to about 1972, this is to be expected. In the study area, land use has remained fairly constant since the 1950's, and much of the area has been extensively developed since the late 1800's.

Generally, there were no large differences in nitrite plus nitrate concentrations with depth, except at site 5, which is near the discharge area. Concentrations of nitrite plus nitrate nitrogen, dissolved oxygen, and dissolved organic carbon (DOC) differed by depth of sample below the water table at this site (fig. 14). Some

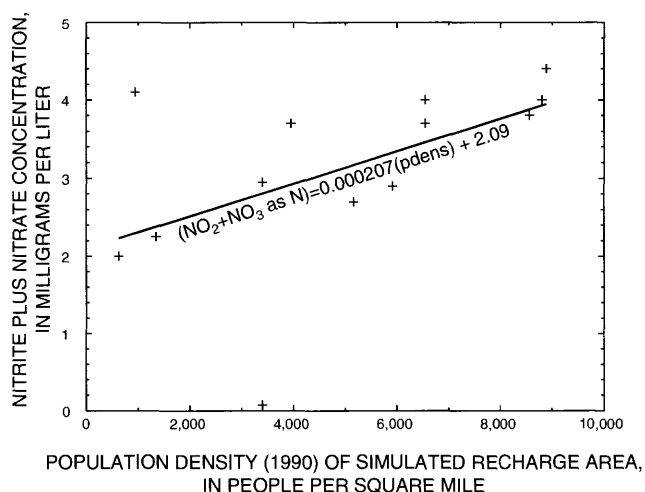


**Figure 14.** Concentrations of nitrite plus nitrate as nitrogen, dissolved oxygen, and dissolved organic carbon by depth of sample below the water table, site 5, Manchester, Connecticut, April 1995.

denitrification possibly has occurred at this location. The low dissolved oxygen and the relatively high DOC concentrations indicate reducing conditions at shallow depths, where nitrate could be reduced to dinitrogen gas through the process of denitrification; however, measurements were not made for these gases. The source of the high DOC concentrations is likely to be organic-rich sediments present in the subsurface at this site.

$\delta^{15}\text{N}$  values for 12 ground-water samples from Manchester ranged from 4.3 to 6.1 per mil. Source areas for most wells were in residential areas with medium- to high-density housing and many maintained lawns; consequently, the principal source of much of the nitrate as nitrogen probably is lawn and garden fertilizers. Some of the nitrate may be derived from leaky sanitary sewer lines, but the isotopic analyses indicate that the nitrate concentrations are not dominated by human and (or) animal waste sources. Other possible sources of nitrate in the Manchester samples include atmospheric deposition and soil-organic nitrogen. Because median nitrite plus nitrate concentrations were much lower in undeveloped areas than in the Manchester flowpath samples (table 9), the atmospheric and soil-derived sources of nitrate could be assumed to be much smaller than fertilizer inputs. However, most of the samples from undeveloped areas were collected in forested areas, where biological uptake of nitrogen may be different than uptake in urban areas; therefore, some contribution from atmospheric and soil-derived nitrate is possible.

Nitrite plus nitrate concentrations are related to population density. Population density of the source areas plotted against the median nitrite plus nitrate concentration for individual wells is shown in figure 15. The linear relation shows that in this setting, estimating concentrations of nitrite plus nitrate in ground water may be possible if the population density is known. A linear-regression model (Helsel and Hirsch, 1992) was used to determine the equation to predict nitrite plus nitrate concentrations based on population density (fig. 15). This relation was statistically significant at the 97-percent confidence level. The relation indicates that for every increase in population density of about 4,800 people/mi<sup>2</sup> in a source area, there is a corresponding increase in nitrite plus nitrate nitrogen concentration of 1 mg/L in ground-water recharge. Samples from two sites did not correlate well. One sample from site 5 (M185) may be



**Figure 15.** Median nitrate concentrations as a function of population density in simulated recharge areas in Manchester, Connecticut.

affected by denitrification; the other sample from site 4 (M179) may be affected by a recharge area that originates in a former agricultural area and may not relate to population density.

The median concentrations of nitrite plus nitrate nitrogen were higher in samples of young water than they were in old water. The higher concentration in young waters could be related to an increase in population density or other factors in the source areas by the time the younger water was recharged to the aquifer.

Nitrite plus nitrate nitrogen concentrations were elevated at most depths and locations in the Hockanum River aquifer. Information on the age of ground water indicates that nitrite plus nitrate concentrations have been elevated in the aquifer for at least the last 23 years. On the basis of the results of the MODFLOW/MODPATH simulation, similarly elevated concentrations will likely continue for many years, even if land-use practices designed to reduce nitrogen inputs to ground water are implemented.

## Trace Elements

One set of samples collected from August to September 1994 was analyzed for concentrations of trace elements. The detections and concentrations of trace elements are summarized in table 11. Trace

elements were not detected at concentrations greater than the USEPA MCLs. Direct comparisons of the trace-element concentrations in the flowpath wells with those from the undeveloped areas were impossible to make because samples were not analyzed for trace elements as part of the NAWQA land-use studies. However, Grady (1994) collected samples for trace elements from different land-use categories in Connecticut and determined that barium concentrations were significantly higher in sewered residential areas than in any of the other land uses studied. Concentrations of barium in the samples collected for this study were likely higher than background concentrations; however, natural sources of barium, probably the mineral barite, are present in the unconsolidated sediments derived from arkosic

bedrock (Krynine, 1950). The concentrations of chromium, copper, nickel, zinc, and uranium detected in this study probably are not greater than ambient concentrations. The one detection of molybdenum may indicate contamination; however, Hem (1985) found that high concentrations of molybdenum are not uncommon in natural waters.

Iron and manganese were analyzed in all seven sets of samples that were collected during the study. The high concentrations of iron and manganese are not uncommon in natural systems. Manganese concentrations greater than 50 µg/L generally are considered unsuitable for public water supplies, and several samples exceeded this concentration during the time period that the samples were collected.

## Pesticides

Samples for pesticide analysis were collected twice in 1994 at all wells. Of the 85 pesticide analytes, only 4 were detected in samples from the shallowest wells at 2 sites (table 12). The pesticides detected include two herbicides (simazine and dichlorprop) and two insecticides (dichloro diphenyl dichloroethylene (DDE) and dieldrin).

Simazine and dichlorprop were detected at site 5 (well M185). Simazine has many agricultural and nonagricultural uses, including lawn applications (Meister Publishing Company, 1991). Dichlorprop is commonly used on road and utility corridors.

DDE, a metabolite of dichloro diphenyl trichloroethane (DDT) also was identified from one sample at site 5. The compound DDT has been banned from use since 1973 but frequently was used in

residential areas and near wetland areas for mosquito control. Dieldrin was detected twice in samples from site 1 (well M189). Dieldrin commonly was used for termite control and for pre-treatment of lumber but has been banned in Connecticut since 1987 for termite control (Mullaney and others, 1992). These detections can be compared with detections in two previous studies, where six

**Table 11.** Detections and concentrations of selected trace elements in samples from 13 flowpath study wells in Manchester, Connecticut

[<, actual value is less than value shown; µg/L, microgram per liter]

Trace element	Number of detections	Concentration range (µg/L)
Barium .....	13	85–340
Chromium.....	13	2–9
Copper .....	5	<1–9
Iron .....	( <sup>1</sup> )	<3–680
Manganese.....	( <sup>1</sup> )	<1–1,000
Molybdenum .....	1	<1–9
Nickel .....	12	<1–5
Zinc.....	8	<1–4
Uranium.....	1	<1–1

<sup>1</sup>Many samples collected as part of major ion analyses.

**Table 12.** Pesticides detected in ground-water samples from Manchester, Connecticut, 1994

[Concentration range: E, estimated. No., number; µg/L, microgram per liter; <, actual value is less than value shown]

Site (fig. 1)	Local well No. (fig. 1)	Pesticide compound or metabolite	Number of detections	Number of wells	Concentration range (µg/L)
5	M 185	Simazine.....	1	1	<0.005–0.006
		Dichloroprop.....	1	1	<0.032–0.14
		p,p', DDE.....	1	1	<0.001–E0.001
1	M 189	Dieldrin.....	2	1	<0.001–0.043

pesticides (atrazine, chlordane, diazinon, DDE, silvex, and DCPA (Dacthal) plus metabolites) were detected in some samples from wells in Manchester (Mullaney and others, 1992; Grady, 1994).

## Volatile Organic Compounds

Samples for VOC analysis were collected three times in 1994 and once in 1995. Of the 14 wells sampled, 12 had detections of at least one compound (reporting limit of 0.2 µg/L). The most commonly detected compound was chloroform, which was present in water samples from 10 wells. The next most commonly detected compound was the gasoline additive methyl *tert*-butyl ether (MTBE). A summary of the detections of VOCs is shown in table 13. Only trichloroethene (TCE), the third most frequently detected compound, was detected at concentrations greater than the USEPA MCL of 5 µg/L at site 5 (well M183).

VOCs were detected throughout the saturated zone at depths ranging from 3 to 60 ft below the water table, confirming that the effects of human activities

extend to at least the bottom of the surficial aquifer. Although general relations with depth could not be discerned, some VOC concentrations varied with depth at individual sites.

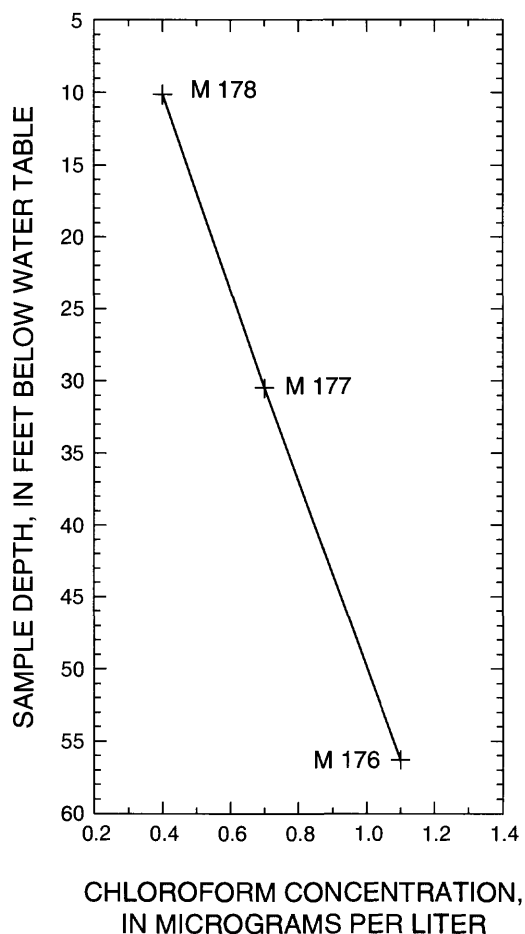
At wells M176-M178 (site 2; fig. 1), chloroform was distributed throughout the sampled thickness of the aquifer (fig. 16). Although differences in chloroform concentration with depth were small, a similar profile was evident during each of the four times these wells were sampled for VOCs. The gasoline additive MTBE also was detected in samples from all three wells at this site.

TCE was detected at two locations in samples from four different wells. At site 2, TCE was detected only in the deepest of the three wells. At site 5, TCE and *cis*-1,2- dichloroethene were detected in water samples from all three depths sampled at this location (fig. 17). TCE concentrations of some samples collected at site 5 exceeded the USEPA MCL of 5 µg/L. The VOC *cis*-1,2-dichloroethene, used in some industrial processes, is a common transformation product of TCE (Pankow and Cherry, 1996). Anaerobic conditions (fig. 17) at some depths at site 5 may favor

**Table 13.** Summary of detections of volatile organic compounds in samples collected from 14 wells in Manchester, Connecticut

[USEPA maximum contaminant level for drinking water: USEPA, U.S. Environmental Protection Agency; P, proposed. <, actual value is less than value shown; p, proposed; --, no standard set]

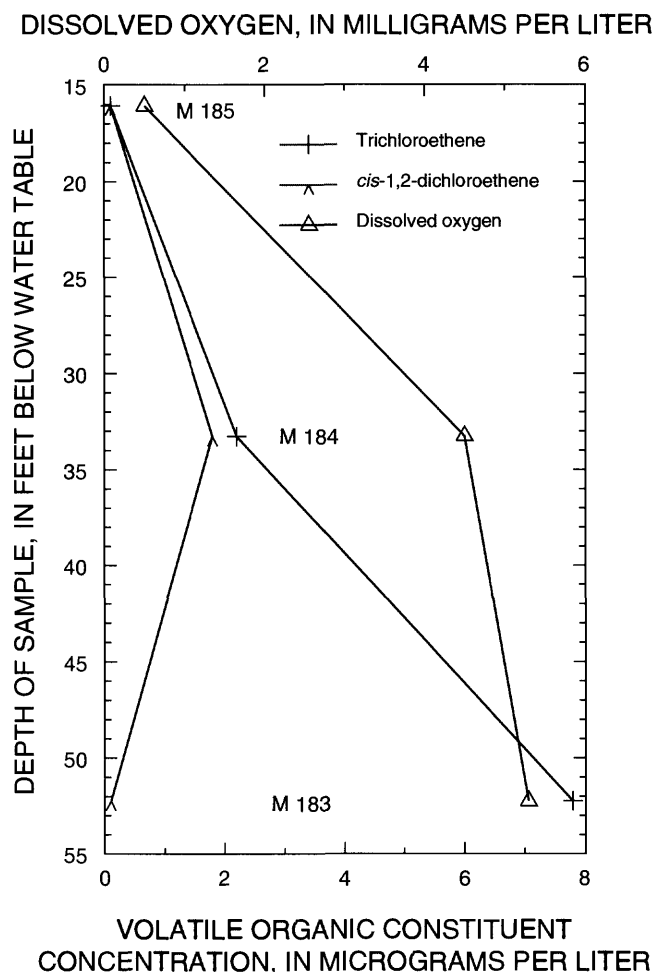
Volatile organic compound	Number of detections	Number of wells	Concentration range, in micrograms per liter	USEPA maximum contaminant level for drinking water in micrograms per liter	Site and local well No. (fig. 1)
Bromodichloromethane .....	1	1	<0.2–0.8	80 p	Site 1--M189
Chloroform .....	33	10	<0.2–13	80 p	Site 1--M189 Site 2--M176, M177, M178 Site 3--M188 Site 4--M179, M180 Site 5--M183, M184, M191
<i>cis</i> -1,2-dichloroethene .....	9	4	<0.2–2.7	70	Site 5--M183, M184, M185, M191
1,1 dichloroethane .....	1	1	<0.2–0.3	--	Site 5--M185
Dichlorodifluoromethane.....	4	1	<0.2–0.7	--	Site 5--M185
Methylene chloride.....	1	1	<0.2–0.8	--	Site 5--M181
Methyl <i>tert</i> -butyl ether (MTBE).....	17	6	<0.2–0.8	--	Site 2--M176, M177, M178 Site 4--M180, M181 Site 5--M185
Tetrachloroethene .....	5	2	<0.2–0.4	5	Site 2--M178 Site 3--M188
Trichloroethene (TCE).....	15	4	<0.2–11	5	Site 2--M176 Site 5--M183, M184, M185



**Figure 16.** Chloroform concentrations as a function of depth of sample below the water table, site 2, Manchester, Connecticut, May 20, 1994.

the transformation. *cis*-1,2-dichloroethene also was detected in the streambed well (M191) at site 5, indicating that this compound may be discharging to the Hockanum River.

There were no general relations between age of ground water and VOC detections or concentrations, except for the compound MTBE, which was detected only in samples of ground water 11 years old or less. The use of MTBE as a fuel oxygenate was mandated by amendments to the Clean Air Act in 1990. Urban areas with air-quality problems, including Connecticut, were required to use oxygenated fuels by November 1, 1992, but MTBE was used in some urban areas since 1988 and in premium gasoline in New England during the 1980's (Squillace and others, 1996). MTBE was



**Figure 17.** Concentrations of trichloroethene, *cis*-1,2-dichloroethene, and dissolved oxygen as a function of depth of sample below the water table, site 5, Manchester, Connecticut, April 13, 1995.

detected in 17 samples from 6 wells at 3 different sites. Five wells had ground-water ages ranging from less than 1 to 6 years, corresponding to the time of widespread introduction of MTBE as a gasoline additive in Connecticut.

Concentrations of VOCs detected more than once in any flowpath well, including chloroform, MTBE, TCE, *cis*-1,2-dichloroethene, tetrachloroethene, and dichlorodifluoromethane, typically changed only slightly with time. Two compounds were present in samples from two different wells at elevated concentrations—chloroform (1.8 to 13 µg/L at site 1) and TCE (7.8 to 11 µg/L at site 5

(M183)). These compounds demonstrated somewhat greater variability in concentration; however, there were not enough samples collected over a sufficient time period to determine any temporal trends.

The overall consistency of VOC concentrations in samples collected from May 1994 through April 1995 was unexpected. In particular, sites with low VOC concentrations were among the most consistent over time. Perhaps this indicates that small amounts of VOCs are being transported to the aquifer from sources that are stationary and relatively constant over time, rather than from random events.

The source of chloroform, the most commonly detected VOC in flowpath wells, is likely the chlorinated water from public-water supplies that reaches the water table through leaks in water mains, sanitary sewer lines, or through lawn irrigation. Concentrations of chloroform in finished water from the Manchester public water supply varies but is usually from 20 to 40  $\mu\text{g/L}$  (Patrick Kearney, Manchester Water Department, oral commun., 1996). Chloroform also can be formed in the aquifer by the reaction of chlorine from public-water and sewer systems with naturally occurring organic carbon. Source areas for flowpaths to sampled wells with chloroform detections have medium- to high-density residential and commercial areas that are served by public-water and sewer systems. Other potential sources of chloroform are associated with manufacturing processes for refrigerants, solvents, and plastics (Smith and others, 1988).

MTBE was detected in samples from six wells at three sites. Low concentrations of MTBE and the lack of detections of other gasoline constituents (such as benzene, toluene, ethylbenzene, or xylene) in ground-water samples collected at these sites suggest that it may be from nonpoint sources. Possible nonpoint sources include small spills of gasoline and subsequent stormwater runoff and infiltration from driveways, parking lots, and streets, and diffusion from the atmosphere through the unsaturated zone to shallow water tables. Because the source areas to all wells where MTBE was detected contain a high percentage of paved surfaces, the runoff from these areas is a likely source. In a USGS study of urban stormwater quality from 1991 to 1995, MTBE was detected in 6.9 percent of 592 samples collected in 16 urban areas (Delzer and others, 1996).

TCE is commonly used as a solvent for dry cleaning and degreasing, in refrigerants, and in fumigants (Smith and others, 1988). The ground-water flowpaths to the wells where TCE was detected move primarily through medium- to high-density sewered residential areas, with some commercial areas. Possible sources include spills or leaks of the compound, as well as leaky sanitary or storm sewer lines. At both sites where TCE was detected, the highest concentrations were on the bottom of the aquifer. Because TCE is a dense non-aqueous phase liquid (DNAPL) (Pankow and Cherry, 1996), a spill or leak possibly could sink through the saturated zone and distribute dissolved concentrations of TCE throughout the vertical profile. The source area of the TCE need not coincide with the recharge area for ground water passing through the well; however, the source would still be somewhere along the flowpath. Also, the age of the water will not necessarily coincide with the time that the contaminant was introduced.

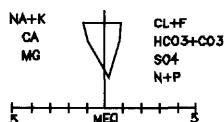
There are many potential sources for solvents such as TCE and some of the other VOCs detected in Manchester. The USEPA collects data on transfer and storage of hazardous chemicals from 76 locations in Manchester to comply with the Resource Conservation and Recovery Act (U.S. Environmental Protection Agency RECRIS database, 1996).

## **EFFECTS OF INTERACTION OF GROUND WATER AND SURFACE WATER ON WATER QUALITY**

Surface-water samples were collected from the Hockanum River upstream and downstream from the study area and at selected locations on Bigelow and Hop Brooks during low-flow periods in 1994 and 1995 to determine the inflow of dissolved constituents from ground water to surface water in the Manchester area. Six samples also were collected from a streambed well (M191) installed in the Hockanum River near site 5 (fig. 1) to measure the concentrations of dissolved constituents entering the stream from ground water at the downgradient end of the study area. The chemical composition of ground water and the low-flow stream samples collected at the downstream end of Bigelow Brook were similar, but the water type of the Hockanum River is different from the ground water and changes from a sodium chloride to a calcium bicarbonate water type as a result of ground-water inflow from the study area (fig. 18).

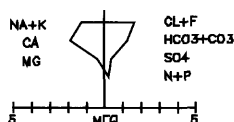
## HOCKANUM RIVER UPSTREAM END OF STUDY AREA

D. Hockanum River upstream  
of study area at Railroad Bridge,  
Manchester, Connecticut



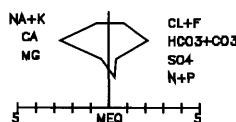
## BIGELOW BROOK

A. Bigelow Brook above Center  
Springs Pond, Manchester,  
Connecticut



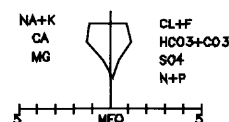
## GROUND-WATER NEAR THE HOCKANUM RIVER

Well M183 at site 5

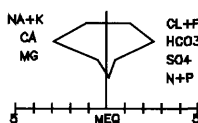


## HOCKANUM RIVER DOWNSTREAM END OF STUDY AREA

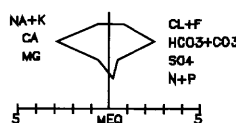
F. Hockanum River  
downstream of study area,  
below New State Road,  
Manchester, Connecticut



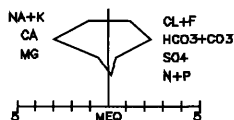
B. Bigelow Brook at Broad Street,  
Manchester, Connecticut



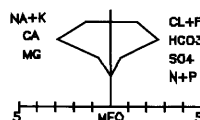
Well M184 at site 5



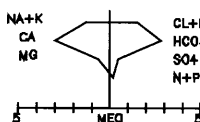
C. Bigelow Brook at  
Hilliardville, Connecticut



Well M185 at site 5



Streambed Well M191 near site 5



**Figure 18.** Concentrations of selected ions at streams and nearby wells in Manchester, Connecticut, August 24, 1995. [NA; sodium, CA; calcium, MG; magnesium, CL; chloride, F; fluoride, HCO<sub>3</sub>; bicarbonate, CO<sub>3</sub>; carbonate, SO<sub>4</sub>; sulfate, N; nitrogen, P; phosphorus, MEQ, milliequivalents per liter]

Mass balance calculations were performed to determine the average concentration of selected inorganic constituents in ground-water inflow to streams between surface-water sampling stations. The values compare well with the concentrations in the ground-water samples (table 14). The upstream instantaneous load was subtracted from the

downstream instantaneous load on each stream to determine the mass of selected constituents from ground water, and that value was divided by the difference in flow. Average concentrations of inorganic constituents in ground-water inflow to the streams also were compared with those from the streambed well that was installed in the Hockanum River, and many of



**Table 14.** Calculated average concentrations of selected constituents in ground-water inflow and median values of ground-water samples collected for this study, Manchester, Connecticut

[mg/L, milligram per liter]

Constituent	Calculated average ground-water inflow concentration			Median concentration in flowpath ground-water samples	Median concentration in streambed well M191
	Surface-water stations A and B Bigelow Brook	Surface-water stations B and C Bigelow Brook	Surface-water stations D and F Hockanum River		
Bicarbonate, field (mg/L as CaCO <sub>3</sub> ) .....	247	150	150	130	170
Calcium (mg/L) .....	81	59	50	52	62
Magnesium (mg/L) .....	8	5.9	5	5	8.4
Sodium, in mg/L .....	22	23	21	18	31
Potassium (mg/L) .....	1.6	1.1	2	1	.9
Sulfate (mg/L) .....	29	23	25	18	22
Chloride (mg/L) .....	32	44	35	38	51
Nitrite plus nitrate as nitrogen (mg/L)....	.37	2.7	2	3.7	2.8

these concentrations also are similar. Concentrations of some constituents in ground-water inflow are possible to estimate in this setting if there is detailed information on ground-water quality. The data also show that the average concentration of nitrite plus nitrate nitrogen in ground water discharging to the stream and the median concentration measured in the streambed well (M 191) were lower than the median ground-water concentrations. This may indicate that some nitrite plus nitrate as nitrogen is being lost, possibly through denitrification, or that water with lower nitrite plus nitrate as nitrogen concentrations is mixing with the ground water. Another possibility is that there is some biological uptake of nitrogen in the surface water.

Information on the quality of ground water discharging to streams in the Manchester area will augment the limited information currently available on ground-water contributions to urban nonpoint-source pollution. Nutrients and other constituents transported in ground water may affect not only the receiving stream but ultimately, the quality of all downstream waters. The instantaneous loads calculated for the low-flow samples from Bigelow Brook and the Hockanum River were compared with instantaneous low-flow loads at a USGS streamflow-gaging station (Hockanum River near Manchester; station 01192500), about 1.5 mi downstream from the study area (site G, fig. 1). This station was sampled monthly during 1993-95 as part of the NAWQA program. The instantaneous loads from

the upstream stations were compared with those collected at the gaging station during the same general time period and at nearly identical flows. The comparison shows that during low-flow conditions, fluxes of water-quality constituents from ground water in the study area can be a large percentage of the total load at the downstream gaging station (site G, fig. 1). For example, the section of Manchester sampled includes only 5.9 percent of the drainage area of the Hockanum River at the gaging station, but during low-flow conditions in August 1995, contributed about 18 percent of the flow, 11 percent of the nitrite plus nitrate nitrogen, 32 percent of the calcium, and 16.1 percent of the chloride (table 15).

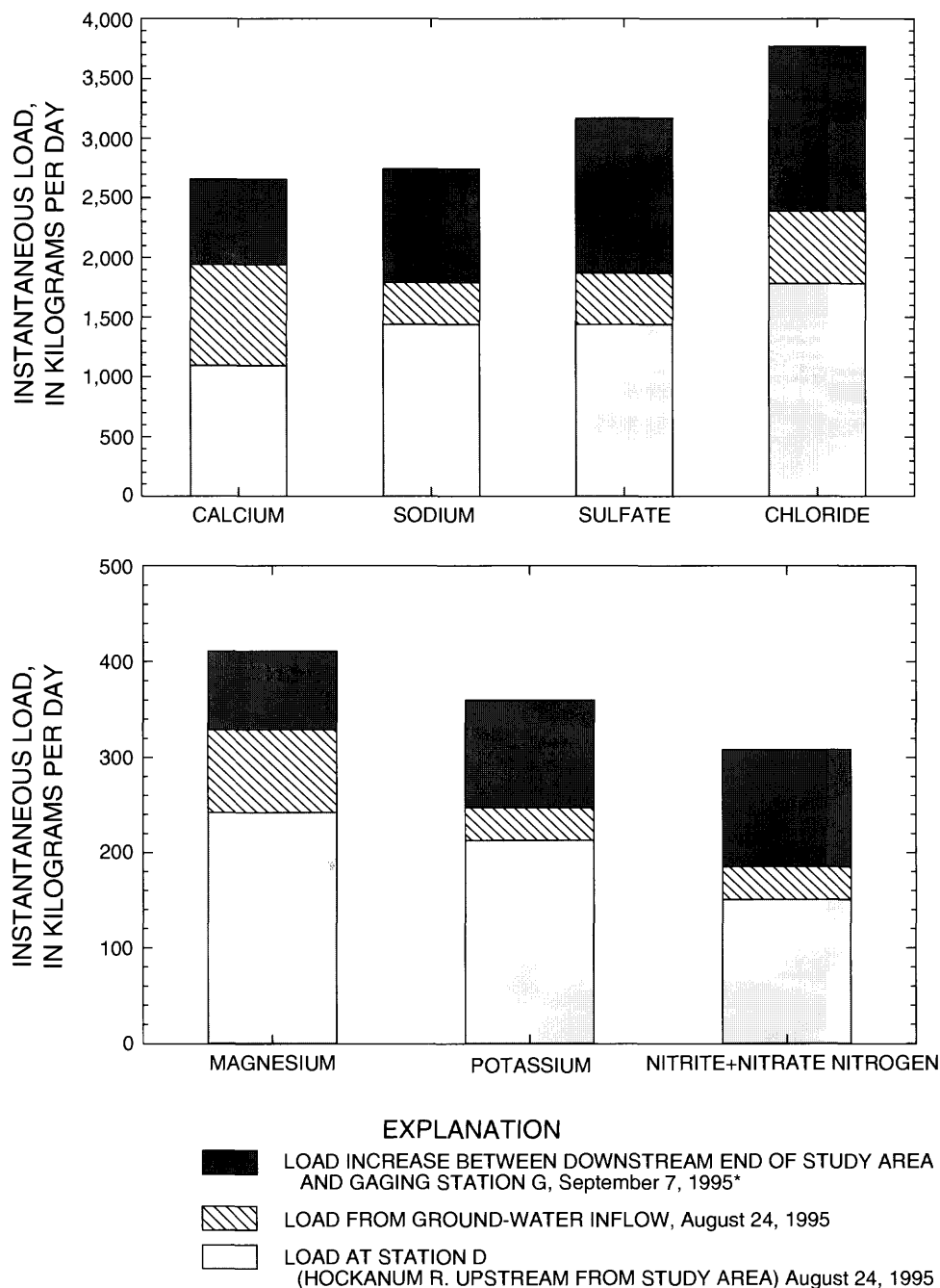
The average daily load of nitrite plus nitrate nitrogen in 1995 for the Hockanum River at site G was calculated to be 680 kg/d (E.C.T. Trench, U.S. Geological Survey, written commun., 1996). A comparison of the ground-water load of this constituent (34 kg/d) during low flow would indicate that at least 5 percent of the load for 1995 could be attributed to ground-water inflow in from Manchester between the two sampled surface-water stations D and F. This is a conservative estimate, because the average baseflow in the streams sampled is likely to be larger than during a low-flow period. Similarly, a large percentage of the loads of other dissolved constituents in the Hockanum River at station G also are derived from ground-water inflow in the Manchester area (fig. 19).

**Table 15. Calculated loads of selected constituents in surface water during low flow, Manchester, Connecticut, July 1994 and August 1995**

[mi<sup>2</sup> square mile; ft<sup>3</sup>/s, cubic foot per second]

Site designation (see fig. 1)	Surface-water sampling location	Drainage area (mi <sup>2</sup> )	Date	Instantaneous discharge (ft <sup>3</sup> /s)	Instantaneous load, converted to kilograms per day						Nitrite plus nitrate as nitrogen
					Calcium	Magnesium	Sodium	Potassium	Sulfate	Chloride	
D	Hockanum River at Railroad Bridge	55.1	8-24-95	23.5	1,090	242	1,440	213	1,440	1,780	151
F	Hockanum River downstream of New State Road	59.5	8-24-95	30.5	1,940	329	1,790	247	1,870	2,390	185
G	Difference stations D to F Hockanum River near Manchester	4.3 73	8-24-95	7.0	850	87	350	34	430	610	34
			7-19-94	59	4,330	750	4,190	590	4,480	5,920	476
			7-21-94	151							
			8-24-95	138							
A	Bigelow Brook above Center Springs Pond	1.4	9-07-95	35	2,660	411	2,740	360	3,170	3,770	308
			8-24-95	.26	24.2	3.2	17.8	0.89	11.5	36	2.0
B	Bigelow Brook at Broad Street	1.8	8-24-95	.49	69.5	7.7	30	1.8	27.6	54	2.2
Difference between stations A and B					45.3	4.5	12.2	.91	16.1	18	.2
C	Bigelow Brook at Hilliarville	3.00	7-21-94	4.2	606	63.7	257	13.4	247	462	26.7
			8-24-95	3.2	461	47.0	180	9.4	180	345	20.4
H	Difference stations B to C Hop Brook near Manchester	.4 11.7	8-24-95	2.71	391.5	39.3	150	7.6	152.4	291	20.2
			7-21-94	2.42	225	24.9	148	10.1	154	266.5	6.5
Percentage of Hockanum River drainage area at gaging station					All values in percent of instantaneous load at the Hockanum River near Manchester gaging station, during same time period with similar low-flow discharge						
Difference D and F as a per- centage of station G					32	21.2	13.0	9.4	13.6	16.1	11
Station C as a percentage of station G					14	8.5	6.1	2.3	5.5	7.8	5.6
					17.4	11.4	6.6	2.6	5.7	9.2	6.6

<sup>1</sup>Mean daily discharge.



**Figure 19.** Low-flow loads of selected constituents from ground water at the Hockanum River, Manchester, Connecticut, August-September 1995. [\* The Hockanum River near Manchester sampled on September 7, 1995, at nearly identical flow conditions.]

A large increase in instantaneous load at low flow also occurs between the most downstream temporary low-flow site on the Hockanum River (station F) and the downstream streamflow-gaging station near Manchester (station G). This increase is

likely due in part to the inflow of Hop Brook and additional ground-water inflow from a section of Manchester that contains the landfill, other urban land uses, and effluent from the Manchester wastewater-treatment plant.

## SUMMARY AND CONCLUSIONS

A study of the hydrogeology and ground-water quality of the Hockanum River aquifer underlying urban areas in Manchester, Connecticut, was conducted in 1993-95 as part of the NAWQA program in the Connecticut, Housatonic and Thames River Basins. This surficial, glacial sand and gravel aquifer is composed largely of sediments that were eroded from the underlying arkosic bedrock and deposited as a series of glacial-deltaic deposits. Horizontal hydraulic gradients typically range from 0.01 to 0.02 ft/ft and indicate a westward flow of ground water towards discharge points along the Hockanum River or Bigelow Brook.

Multiple samples were collected over a 2-year period from 14 flowpath wells. The wells were primarily installed in clusters of three and are screened in shallow, intermediate, and deep parts of the aquifer along the direction of ground-water flow. Samples were analyzed for major ions, field parameters, nutrients, trace elements, DOC, pesticides, and VOCs. In addition, samples were collected in June 1995 to determine the age of ground water by analysis of  $^3\text{H}$ - $^3\text{He}$  concentrations. The ground water ranged in age from 0.9 to 22.6 years. This information was used to calibrate a ground-water-flow model and to interpret ground-water-quality analyses.

A steady-state, single-layer, finite-difference ground-water-flow model was used to improve the understanding of the ground-water system and estimate the travel distance from source areas to the sampled wells. The model indicated that travel distances from recharge areas to wells ranged from about 50 to 11,000 ft. A particle-tracking analysis indicated that flowpaths were longer from source areas to points on the bottom of the 5-foot well screens than to the top of the well screens. The difference in travel distance ranged from several hundred to several thousand feet and was largest for wells screened in the deepest part of the aquifer. This information indicates that the ground-water samples collected from wells in this study may represent more than 1 year of ground-water recharge.

Land use in source areas to the sampled wells was determined in zones delineated by the ground-water-flow simulation and includes at least some high-density residential and commercial areas and medium-density residential areas. Population density also was determined in the source area to each well and ranged from 629 to 8,895 people/mi<sup>2</sup>. A conclusion of this study is that urban land use does have an effect on

ground-water quality in the Manchester area, but there are many complex natural and human factors that contribute to water-quality conditions. These factors include land use in the source area, population density, ground water traveltime, location and direction of flowpaths through the saturated zone, climate, and variations in constituent input with time.

Concentrations of selected inorganic constituents did not exceed primary drinking-water standards. Median concentrations of sodium, chloride, nitrite plus nitrate as nitrogen, bicarbonate, calcium, and dissolved solids from shallow flowpath wells were higher than concentrations in samples from wells in undeveloped areas in Connecticut with similar geochemical characteristics. The most probable source of high sodium and chloride concentrations is runoff of deicing chemicals from roads and parking areas. All source areas contain a large number of streets, driveways, and parking areas. The most likely source for high concentrations of nitrite plus nitrate as nitrogen (as determined by stable nitrogen isotope ratios) is fertilizer applications to lawns and gardens, but other sources, such as atmospheric deposition, natural soil nitrate, and leaky sewers, also may have some input. Source areas to all wells contain sewered areas and high- and medium-density residential areas with lawns, and population density in source areas was determined to be a factor in nitrite plus nitrate as nitrogen concentrations.

In samples from the flowpath wells, values of some inorganic constituents, including sodium, chloride, and selected field parameters, differed by depth of sample below the water table. In general, most constituents did not differ greatly with depth of sample. Concentrations of nitrite plus nitrate as nitrogen were similar with depth in the aquifer, except at site 5 (the discharge end of the aquifer), where the concentration of nitrite plus nitrate as nitrogen may be reduced by denitrification.

Median concentrations of dissolved oxygen were lower and pH and concentrations of bicarbonate, alkalinity, total hardness and calcium were higher in old ground water (10-22.6 years) than in young ground water (0 to 10 years). These differences likely reflect a longer interaction with geologic materials. Median concentrations of sodium, chloride, and nitrite plus nitrate as nitrogen were higher in samples of young ground water than in samples of old ground water. These differences are likely caused by anthropogenic

inputs and may relate to differences in input with time (the overall input was larger in the last 10 years), or differences in land use in the source areas to the wells.

Concentrations of many inorganic water-quality constituents at individual wells differed by 10 to 40 percent with time. There were an insufficient number of samples over a sufficient period of time to determine many seasonal or temporal patterns in the data. Shallow wells with the largest water-level fluctuations showed more frequent variations in concentration of inorganic constituents and field parameters. Because the minimum age of water at any of the wells was 0.6 years, and because the sampled waters are likely to be a mixture of several years of recharge, variations in concentration are difficult to relate to events on the surface, even though some differences (such as in chloride concentration) were observed. Point samples collected at the water table may be useful to determine variations in input from surface sources. Variations in time that occur in response to short-term minor changes in ground-water flowpaths caused by water-level fluctuations also are difficult to separate from those caused by variations in input of anthropogenic sources.

Samples were collected and analyzed for trace elements, and no concentrations were detected at concentrations greater than the established USEPA MCLs for drinking water. With the exception of barium, concentrations of trace elements probably are not elevated as compared with ambient conditions. Four pesticides, including two insecticides (DDE, a metabolite of DDT, and dieldrin) and two herbicides (dichloroprop and simazine) were detected, and concentrations were low.

Nine VOCs were detected in samples from the flowpath wells sampled for this study. The four most commonly detected VOCs were chloroform (10 wells), MTBE (6 wells), TCE (4 wells), and *cis*-1,2-dichloroethene (4 wells). Only trichloroethene was detected at concentrations greater than the USEPA MCL for drinking water of 5 µg/L. VOCs were detected at depths from 3 to 60 ft below the water table, demonstrating that the effects of human activities extend to the bottom of the surficial aquifer in Manchester. Several VOCs were detected in samples of both old and young ground water, but MTBE was detected only in samples that had ground-water ages of 11 years or less, which corresponds to the time of introduction of MTBE to gasoline supplies in

Connecticut. The source of MTBE in the samples from six of the flowpath wells is likely from nonpoint sources, including runoff and infiltration from parking lots and streets and atmospheric sources. The source of chloroform, detected in 10 of 14 wells, is likely the chlorinated water from public-water supplies that reaches the water table through leaky water mains, sanitary sewer lines, or through lawn irrigation. The exact sources of the detected VOCs are not evident, but there are many locations in Manchester where many VOCs may have been used.

Ground-water samples were collected from a well installed in the streambed of the Hockanum River, and surface-water samples were collected at six different sites during low-flow periods in 1994 and 1995 to determine the amount of dissolved constituents entering surface water from ground-water discharge. A comparison of water type showed that the composition of ground water near the discharge area was similar to the composition of surface water at low-flow conditions. Average concentrations of selected water-quality constituents in ground-water discharge between stations on Bigelow Brook and the Hockanum River were compared with median concentrations in all ground-water samples and in the streambed well. This comparison indicates the measurements of ambient ground-water quality can be used to estimate potential effects of ground-water discharge on surface-water quality. The comparison also showed that the concentration of nitrite plus nitrate as nitrogen entering the stream from ground water was lower than the median concentrations in ground water, possibly due to denitrification in the zone adjacent to the river or biological uptake of nitrogen in the river.

Instantaneous ground-water loads of inorganic constituents were calculated on the basis of the low-flow sampling at two stations on the Hockanum River and compared with loads determined from a downstream streamflow-gaging station where monthly samples were collected. During periods of low streamflow, ground-water discharge from the Manchester flowpath study area may contribute 11 percent of the nitrite plus nitrate as nitrogen, 32 percent of the calcium, and 16 percent of the chloride to the Hockanum River. A comparison with daily load calculations for 1995 indicates that ground water in the study area likely contributes at least 5 percent of the annual load of nitrite plus nitrate as nitrogen at the downstream gaging station.

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